Drivers of spatial and temporal variability in the fire regime of boreal forest in western Russia

Abstract: Fire is an integral and major control on forest properties and ecosystem functions within the boreal forest. However, significant spatial and temporal variability in fire regimes occurs across this area that may not current be being encompassed in models and predictions of its future, particularly under a changing climate. This study has attempted to determine the relative controls on spatial and temporal variability in fire regime within the less well-studied region of western Russia, over the past two decades. While results show some similarity to other regions of Russia, there is an inherent and significant variability within this system that must be accounted for. The interaction between anthropogenic and natural drivers seems to be particularly important here. Scaling this up across the boreal forests is crucial if we are to understand how fire disturbance regimes will respond to amplified warming in high latitude regions.

Introduction

The boreal forest is the largest biome in the world, spanning approximately 1.2 billion hectares in a circumpolar belt that stretches across Eurasia and North America (Stocks & Lynham, 1996). Its northern and southern boundaries are typically delineated by the 13 °C mean July isotherm in the north and the 18 °C mean July isotherm in the south, roughly situating it between 45 and 70° north latitude (Larsen, 1980). The complex interaction between extremes in climate, solar radiation, topography, nutrient availability and disturbance regimes that occur within these latitudes gives rise to forest that is characterised by floristically simple but hardy vegetation (Soja et al., 2007). Hence the boreal forest is largely composed of just eight genera, which vary in dominance across its vast range: needleleaf pine (*Pinus*), larch (*Larix*), spruce (*Picea*) and fir (*Abie*s), and deciduous birch (*Betula*), aspen (*Populus*), willow (*Salix*) and alder (*Alnus*) (Brandt 2009). Despite this apparent simplicity, the boreal forest is of extreme global and regional importance. In particular, the vast area of the forest and the unique cold-weather interactions present within it mean the boreal region plays a major role in the climate system (Bonan, 2008). For example, the cool organic soils here are the largest global store of terrestrial carbon (Pan et al., 2011), while the region represents some 30% of global forest cover that has significant albedo feedback (Kuusinen et al., 2012). With around two-thirds of the boreal forest located in Eurasia, and the remaining third located primarily in Canada and Alaska (Larsen, 1980), it is also a major resource for local populations and supports a number of industries within the countries it spans (Ruckstuhl et al., 2008). However, while the extremity of the boreal forest makes it a unique and important ecosystem, it also renders it particularly vulnerable to anthropogenic-induced climate change (e.g. Soja et al., 2007; Bonan, 2008). Arctic amplification of mean global warming trends is predicted to almost double the average magnitude of winter and autumn warming experienced at high northerly latitudes (Serreze & Barry, 2011). This will impact a number of ecological processes within the Boreal forest system, with largely positive feedbacks to increase climate warming (e.g. Conard & Ivanova, 1997; Goetz et al., 2007; Kuusinen et al., 2012; Tei & Sugimoto et al., 2018). A particularly major component of the system that will be significantly impacted from climate change is wildfire disturbance (Shuman et al., 2017).

Boreal Forest Fire

Fire is an integral component of the boreal forest system. A natural phenomenon that occurs at yearly to centennial intervals, it is crucial for maintaining forest health and functioning (Furyaev et al., 2001). Namely, fires initiate the process of regeneration in the boreal forest, speeding up vegetation succession and nutrient cycling to play a key role in controlling forest development (Burkle et al., 2015). They are both a function-of and control-on this ecosystem, in particular a major determinant of stand composition and structure (Furyeav et al., 2001). This, in in turn, is key in influencing fire behaviour (e.g. Krylov et al., 2014). Hence, while fires are a dominant disturbance agent across the boreal forest, fire occurrence and the effects of fire are not homogenous (Conard & Ivanova, 1997). External influences are also major drivers of fire patterns. Logically, climate is a particularly strong external control on fire disturbance, shaping where and when fires occur (Soja et al., 2007). Anthropogenic climate warming that extends the fire season, reduces intervals between fire occurrence and increases conditions of extreme-fire weather (hot, dry) is therefore predicted to result in considerable increases in the fire frequency and area burned across the boreal forest (e.g. Kasischke & Turetaky, 2006; Flannigan et al., 2009; Strahlberg et al., 2018). The overall effect and potential climate feedbacks of this are unclear, however (Goetz et al., 2007). For example, increases in area burned are likely to increase the amount of carbon released from burning vegetation and soils, initiating positive feedback on warming (Randerson et al., 2006; Kukavskaya et al., 2012). On the other hand, post-fire regenerating stands have enhanced productivity, sequestering more carbon than old growth areas, as well as significantly altered albedo (e.g. Jin et al., 2012). The differences lie in whether fires will become more frequent, more severe, or both, or be attenuated by some threshold beyond which fire activity destabilises this pattern to limit any further increase (Shvidenko & Schepaschenko, 2013). Given that a number of factors influence fire, including forest properties and human activity, understanding the effects of climate-induced change on fire in the boreal forest therefore requires an understanding of current spatial and temporal variation of fire in this system.

Fire Regime

Fire disturbance in a forest area is composed of two parameters: i) *fire frequency* – the number of fires per unit area, and ii) *fire intensity* – energy emitted per unit area (Keeley, 2009). Together, these determine *fire severity* – or how significant fire is in influencing ecosystem processes. It is important to note that while fire severity is often used interchangeably with fire intensity, they are different. Fire intensity is a direct or indirect measure of fire energy output; fire severity refers to metrics which indicate how severely a particular ecosystem component has been effected by fire, they not a measure of the fire itself (Keeley, 2009). Fire frequency and intensity are influenced by a range of factors that alter over space and time, including climate, weather (e.g. wind), topography, forest stand structure and composition, and previous fire activity (Shvidenko & Nilsson, 2000; Gralewicz, 2008). The severity of fires therefore also varies over space and time. The *fire regime* of a particular area is the long-term aggregation of this spatial and temporal variability, determining how systems are affected by fire (Shvidenko & Nilsson, 2000). Broadly, fire regimes can be categorized along a spectrum of high frequency *or* high severity. That is, the effect of either parameter being consistently high generates negative feedback to prevent extremes in the other (e.g. Johnstone et al., 2010). Hence, while particularly intense and frequent fires may occur in some areas some of the time, high frequency, high intensity fire *regimes* are not found within natural systems. Theoretically, the effect of such regimes would be so destabilizing as to simply prevent long-term occurrence in a particular area (Goetz et al., 2007). Regimes can also be divided by the predominant behavior of fires within a forest stand. Low-intensity fires are commonly classed as 'surface fires' – where little fire reaches vertically into the forest canopy. These can be non-stand or partially stand-replacing, but do not result in total loss of forest cover (Krylov et al., 2014). In contrast, high-intensity fires often occur as 'canopy' fires – flames reach high into the canopy, causing partial or total stand-replacement as forest cover is removed (Krylov et al., 2014).

Fire Regime in Russian Boreal Forests

The fire regime of Russian boreal forests is generally regarded as being dominated by high frequency, low-intensity surface fires (e.g. Korovin, 1996; Gromtsev, 2002; Shuman et al., 2017). This is supported by a fairly large body of work on fire statistics and behavior, summarized in Table 1. Generally, the research finds that species composition and stand structure is a major determinant of dominant fire regime in the study areas (e.g. Krylov et al., 2014). On the other hand it is climate, namely sustained drier and hotter than average conditions, that is the main reason shifts in dominant regime occur (e.g. Kajii et al., 2002). Interactions of a warmer climate with other forest-level properties are the mechanisms by which current fire regimes are thought to differ from those indicated in reconstructions (e.g. Kharuk et al., 2011). However, it can be seen that the majority of these studies have been based in eastern and central portions of Russia, particularly Siberia (Table 1). While the

Authors	Study Area	Fire Statistics	Fire Behaviour
Ivanova et al., 2010	Tuva, south-central Siberia	212 fires per year 0.28×10^5 ha burned per year	Surface - canopy fires during droughts
Krylov et al., 2014	Russian Federation	19.66×10^5 ha burned per year 41.66×10^5 ha burned per year	North – canopy South – surface
Chen & Loboda, 2018	Southeastern Siberia	11.13×10^5 ha burned per year (non-stand replacing) 9.405×10^5 ha burned per year (stand-replacing)	Majority non-stand replacing (low-moderate intensity)
deGroot et al., 2013	Central Siberia	1141.9 fires per 100 million ha of land 27×10^5 ha burned per year	Over 90% surface (low-intensity) fires
Kukavskaya et al., 2013	Siberia	813 fires per year 62.5×10^5 ha burned per year (upper estimate of various satellite data)	50% surface fires
Korovin & Isreav, 1998	Russian Federation – Russian Forest Fund	10 - 30,000 registered fires $5 – 21 \times 10^5$ ha burned per year 50-200 ha average fire size	80% surface fires
Wu et al., 2018	Biodiversity hotspots – Far East, south Siberia, North Caucasus	194 ha average fire size (European Russia) 38 ha average fire size (Asian Russia)	Frequent Extreme fire years observed
Kajii et al., 2002	Siberia and northern Mongolia	110×10^5 ha burned (one-year study only)	Strongly climate controlled
Kharuk et al., 2011	Central Siberia	Paleo-record reconstruction: long fire return intervals	50% stand replacing, 50% non-stand replacing Significant effect of topography
Soja et al., 2004	Siberia	$0.234\text{-}13.3 \times 10^5$ ha burned per year	Annual and inter-annual variability Underestimation of area burned

Table 1. Summary of the existing work on boreal forest fire regimes.

central-eastern forest is often studied as it is the largest contiguous area of relatively undisturbed forest cover within the Russian Federation (Brandt, 2009), generalizing findings from limited sites within this could be misleading. This is because western forests differ substantially to central and eastern regions in forest species composition and the level of anthropogenic activity they experience (e.g. Conard & Ivanova, 1997; Mollicone et al., 2006). If forest type and structure are major determinants of fire regime, different dominant vegetation between these areas could be a potential source of significant variation in fire statistics and behaviour across Russia. Eastern forests, particularly those of Siberia, are dominated by larch (*Larix spp.*) and Scots pine (*Pinus sylvestris*) (Kharuk et al., 2011). To the west, however, larch gives way to birch (*Betula spp.*) and spruce (*Picea spp.*) overtakes pine (Brandt, 2009). Pine and larch are generally classed as fire 'resistant' species, with adaptations to fire including thick bark that prevents damage in low severity events; persistent drop of low-hanging branches that limits development of fuel ladders; and relatively low canopy closure that reduces likelihood of canopy fires (Wirth, 2005). These adaptations result in fire regimes dominated by low-intensity but high frequency surface events, as found within the existing literature (e.g. Kharuk et al., 2011). In contrast, the dense, closed canopies of spruce forests increase fuel loading to promote the occurrence of severe canopy fires (Wirth, 2005). Classed as a fire 'avoider' species, spruce has extremely low resistance to fire, meaning they can result in significant tree mortality and stand replacement (Wirth, 2005; Gromtsev, 2002). Hence, fires in spruce forests are extremely rare, occurring on centennial timescales (Gromtsev, 2002). While birch can withstand low-intensity fires, its ability to dominate the boreal forest fire-mediated system generally occurs due to its competitive ability in post-fire successional stages, hence it is often classed as a post-fire 'invader' species (Wirth, 2005). Dominant species in the west therefore do not necessarily support the same regimes as those of eastern forests, particularly if dominant regime is indeed controlled most strongly by forest properties. Such variation in fire regime, if it exists, may therefore not be being captured in current climate-vegetation models that are based on generalization across Russia (e,g. Crevoisier et al., 2007). This may limit the accuracy and reliability of predictions on the effects and feedbacks of climate change here (Krylov et al., 2014). Encompassing various spatial patterns of fire activity across the Russian boreal forest in general climate circulation models (GCMs) has been demonstrated to generate a fourfold difference in carbon release under different climate scenarios, for example (Conard & Ivanova, 1997). It is well documented in eastern forests normally subjected to frequent, less-intense fire regimes, that when larger, more-intense fires do occur they have a much more severe impact, with significant forest loss, carbon release and slow recovery post-fire (e.g. Ivanova et al., 2010). A limited understanding of fire regime in the west means we could therefore be under or over estimating the sensitivity of the region to these events. It is clear that to generate accurate predictions of the effects of fire on forests, including their potential feedbacks climate change, a better understanding of spatial variability in fire regime across Russia is required (Shuman et al., 2017).

Furthermore, western Russia also varies considerably in the amount of human influence it experiences compared to the rest of the country. Namely, while it comprises approximately a quarter of the Russian Federation administrative land area, it contains approximately 70% of the total population (Potapov et al., 2011). Extension of human activity into the forest, through settlements, transport networks, agriculture and extractive industries, is therefore likely to cause variation in the fire regime experienced by this region (Ruckstuhl et al., 2008; Potapov et al., 2011). Forest management here can include the suppression of wildfires where they endanger lives and livelihoods (Malysheva, 2004). Hence, this may reduce the frequency or size of fires here compared to further east. In contrast, human activity may increase the frequency, size or severity of fires by providing additional sources of ignition and altering physical landscape properties to make conditions more suitable for fire (Mollicone et al. 2006). Research from intensively-managed boreal forests in Canada has shown that the dominant force tends to be the latter, with anthropogenic presence largely increasing fire frequency due to purposeful or accidental ignition by people (Campos-Ruiz et al., 2018). Few studies have examined the relative importance of human activity in regulating fire regime within Russian boreal forest, however. In particular, whether this can become a dominant control on fire statistics and behavior and hence regulate the response of fire regimes to climate change. While this is important for accurate predictions of the response of the boreal forest to change, it is also extremely necessary for successful forest management and conservation. For example, allowing identification of areas where human activity exacerbates wildfire to enable the activity to be regulated or the area protected to limit the spread and ecological impacts of fire (e.g. Wu et al., 2018). Theoretically, therefore there is not only likely to be spatial variation in fire regime across Russia, but variation with certain regions. Understanding this at regional levels is important for managing and conserving this sensitive but vital ecosystem (Flannigan et al., 2009).

Fire regime in the boreal forest is likely to be majorly affected by climate (e.g. Kasischke & Turestky, 2006; Johnston et al., 2010). As mean shifts in temperature and precipitation conditions are observed due to anthropogenic climate change, temporal variation statistics and behavior of fires can also be expected (Shuman et al., 2017). Modeled predictions on changes to fire across the boreal forest have already been matched by observations. For example, Soja et al., (2007), found that extreme fire seasons occurred in seven out of nine years between 1998 and 2006 across the boreal region, while other studies within Russia have recorded increases in burned areas in the most recent decades (2000 onwards) compared to the 1990s and decades prior to this (e.g. Ivanova et al., 2010; Kharuk et al., 2011; Shuman et al., 2017). However, understanding temporal variation in fire regime in Russian boreal forest is particularly difficult given a lack of historic data with which to compare current measurements (Soja et al., 2007). The records that do exist from the Russian Federal Forest Service (RFFS) must also be used with caution given they only monitor burns on protected land and in the past have been subject to economic and political pressures that have affected the consistency of reporting between decades (Shvidenko & Nilsson, 2000). Nonetheless, it is clearly important to identify the accuracy of model predictions on the effects of climate change on boreal forests, especially due to the significant and major feedbacks this biome could have (Bonan, 2008; Shvidenko & Schepaschenko, 2014). Due to an overall lack of data from western Russia, it is again unclear what the current temporal trends are in this region, and if they are indeed being mediated by other factors not as present within the rest of the country. Studies utilizing both remote sensing and ground-based observations are sorely needed (Kukavskaya et al., 2013; Wu et al., 2018).

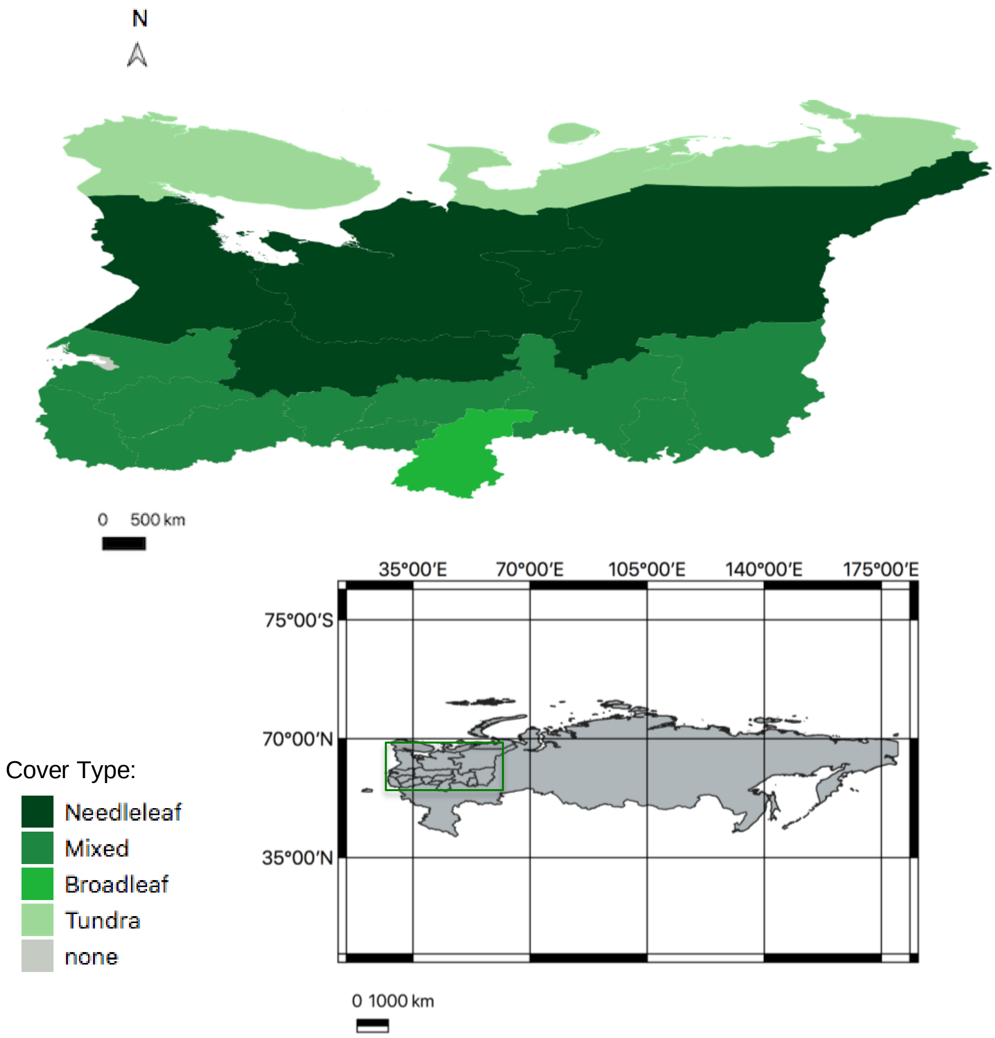
In light of this, the aims of this study were three-fold:

1. To characterize the fire regime of boreal forest in western Russia according to the frequency, intensity and severity of fires
2. To determine the relative influence of forest and external factors in shaping fire regime within western Russian boreal forest
3. To determine the effect of climate change on western boreal forest fire regime over the past two decades.

Materials and Methods

1. Study Area

The study area covers approximately 226 million hectares in northwestern European Russia (Figure 1). It is composed of eighteen administrative districts across the full extent of boreal forest cover in this region, bordered in the south by cropland and steppe, and to the east by the Ural mountains. Information on land cover and species composition of the forest was obtained from the Russian Academy of Sciences' Space Research Institute VEGA-Science database, a collective service based on long-term archives of satellite data and information across northern Eurasia. Four main cover types were classed from the VEGA data, broadly divided according to relative dominance of the main genera present. These are: broadleaf forest, mixed forest, needleleaf forest, and tundra (Figure 1). To ensure fire statistics were pertinent to the boreal forest, total forested area across the study area was calculated and stratified by cover type using the VEGA interactive map tool, giving an overall forest cover of nearly 194 million hectares. The cover types also show spatial separation according level of human activity, highest in broadleaf due to high-density agriculture, and lowest in the tundra with a much lower population.



Cover Type	Total forested area (ha)	Genera (most-least dominant)
Broadleaf	4.05×10^8	*Betula spp.* (Birch) *Quercus spp.* (Oak) *Populus spp.* (Aspen) *Pinus spp.* (Pine)
Mixed	52.8×10^8	*Pinus spp.* *Betula spp.* *Populus spp.*
Needleleaf	110×10^8	*Pinus spp.* *Betula spp.* *Picea spp.* (Spruce)
Tundra (sub-arctic forest-tundra)	19.5×10^8	*Pinus spp.* *Picea spp.* *Betula spp.* (+ gramminoid shrubs & grasses)

Figure 1. Study area of western Russia and characteristics of the identified dominant land cover types across this region.

2. Fire Regime

The fire regime of the study area was assessed between 2001 and 2018 using data on fire frequency, intensity and severity from Landsat series imagery and the MODIS MCD64A1 Burned Area product. MODIS burned area data was obtained from the University of Maryland Fire Information for Resource Management System. The MCD64A1 product identifies burns using active and post-burn fire detection (Giglio et al., 2018). The former method utilizes the distinct mid-infrared spectral signature of fires actively burning as the satellite passes overhead to recognize fires, while the latter uses a fire-sensitive vegetation index in the near and short-wave infrared wavelengths to identify fires based on post-burn changes to vegetation reflectance (Giglio et al., 2018). The combination of methods in this algorithm helps to identify and remove false burns (i.e phenomena which have similar spectral signatures to burning fires or fire-affected vegetation) (Giglio et al., 2018). MODIS was used as it provides extensive temporal coverage with high repeat frequency for the study period. It is also especially effective at identifying small fires, which previous studies indicate could be quite frequent in this area. The first year for which data was available for the duration of the fire season (February – October) was 2001, with data then available for all successive years up to 2018. The data was formatted and processed in QGIS, an open source GIS application, to obtain information on the number of fires, size of individual fires, total area burned, and to identify the timing of burns throughout the fire season. The size of fires has here been used to indicate fire intensity (e.g. per Keeley, 2009). Burns not occurring within the forested region of the study area (e.g. in cropland or cities) were removed and not included in the dataset. For the fragmented forest in broadleaf, this required higher spatial resolution Landsat imagery to identify cropland burns. However, this was not available for all years of the study.

Imagery from the Landsat 4-5 Thematic Mapper, Landsat 7 Enhanced Thematic Mapper Plus and from the current Landsat 8 Operational Land Imager was obtained for 2000-2018 from the USGS (United States Geological Survey) Earth Explorer application and used to derive fire burn severity. Given the large time frame and spatial extent of this study, thirteen locations across the study area were selected to examine using high-resolution imagery, with the number for each type stratified according to forested area (Figure 2).

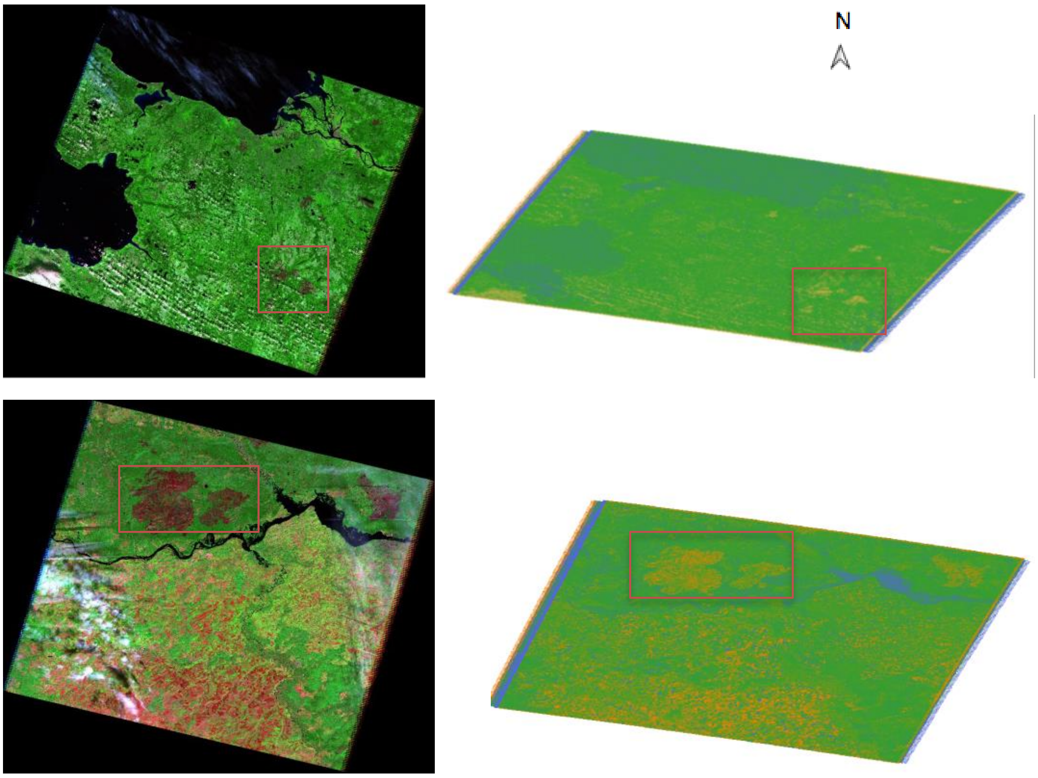
Figure 2. Locations of Landsat image tiles used to derive fire burn severity from NBR. Naming convention is as follows: Tundra = two locations, “Tundra 2” in west and “Tundra 1” in the east; Needleleaf = six locations, “Needleleaf 1-6” from west to east; Mixed = four locations, “Mixed 1-4” from east to west. Broadleaf = 1 location.

For each of the selected locations, it was attempted to find an image with less than 10% cloud cover at the beginning (February-April) and end of the fire season (September-October) for each year. Where this was not possible, an image was selected mid-season; if this was not possible then adjacent image tiles were selected to at least partly cover the area. Across all locations there were, however, some years where no suitable images were available. In total therefore, 178 images were used. They were first viewed in natural colour and false-colour infrared in Multispec to identify burns and verify MODIS burned area against these (Figure 3). Images were then processed in QGIS for top of atmosphere reflectance values before Normalised Burn Ratio (NBR) was calculated for each of pre and post-season tiles in each year according to equation one (Soverel et al., 2010). dNBR could then be calculated according to equation 1a (Soverel et al., 2010).

Eq 1: NBR = (Near Infra Red (NIR) – Shortwave InfraRed (SWIR)/(NIR + SWIR)

Eq 1a: dNBR = $\text{PrefireNBR}_{(year)} - \text{PostfireNBR}_{(year)}$

Normalised Burn Ratio was specifically developed for use in Landsat band wavelengths to assess vegetation recovery after fire (Formacca et al., 2018). It gives values on a scale of -1 to +1 that indicates how severely vegetation has been burned (Figure 3).



dNBR value	Burn Severity Classification
-0.1 - < -0.25	Post-fire regrowth
-0.1 to +0.1	Unburned
0.1-0.44	Low-moderate severity burn
0.44-0.66	Moderate-high severity burn
> 0.66	High severity burn

Figure 3. Identification process of burns from Landsat imagery, with burn scars in red boxes. First viewed in false-colour infrared in Multispec (left image, black border) with dNBR for the burn then calculated (right image) and burn severity classed according to the key (bottom).

It has been used in this study because it contrasts reflectance in wavelengths that are particularly sensitive to fire - the near and short-wave infrared. Because of this, it is much more effective at detecting vegetation change due to fire than other indices, such as Normalised Difference Vegetation Index (NDVI), and it also more useful for assessing vegetation recovery post-fire (e.g. Hislop et al., 2018). The dNBR value of pixels under each MODIS burn polygon was extracted, and mean dNBR for the location was calculated from the mean of values across all fires within the image tile. Burn severity was then classed by dNBR value using USGS NBR standard classification (see Figure 3).

3. Natural and Anthropogenic Controls

To determine the relative influence of potential controls on fire regime across the study area and duration, additional data on climate and human influence was acquired.

Monthly temperature anomalies from the European Centre for Medium Range Weather Forecast (ECMWF) ERA-Interim reanalysis dataset were used to indicate higher-level atmospheric forcing, accessed from the Copernicus Climate Data Store. The dataset is a re-analysis from 1979 to the present, using information from meteorological observations and forecast modeling (ECMWF, 2019). Monthly surface air temperature anomalies are measured between -6 to +6°C of departure from the climatological averaging period between 1981-2010 (ECMWF, 2019). The gridded data files were processed in QGIS and cut to the study area. Spatial and temporal correlation with fires in each month was then assessed.

The data on roads across the study area was accessed from DIVA-GIS, an open source platform of global maps and data. The file was composed from data sourced from the Digital Chart of the World, a comprehensive freely available information source. However, this has not been updated since 1992 and it is likely that there may now be more roads across the study area than shown in this. The distance between roads and fires was calculated in QGIS by the distance from each fire to the closest point on the nearest road. For values of frequency, the distance of each fire to the road was rounded to the nearest kilometer and totaled to give the frequency for that distance.

4. Statistical Analysis

All statistical analysis was carried out in R. Namely, temporal and spatial differences were assessed between years and cover types using one-way ANOVA on respective means, while the relationship between fire frequency and size and proximity to roads was assessed using linear regression.

Results

1. Fire Regime

i. Fire Frequency

In total, 17,616 fires occurred across the study area during 2001-2018, with a mean annual frequency of approximately 78 fires per 1 million hectares of land. There is good agreement between peaks in fire activity and total area burned for each year (Figure 5). There is no significant difference in mean number of fires or area burned between each decade covered in this analysis (Welch t-test, $p > 0.05$). There was, however, considerable temporal and spatial variation in activity during the study period.

MODIS burned area data shows that there areas where fire activity is high across all years, and areas where fire activity is extremely low (Figure 3). In particular, high fire activity occurs in the broadleaf district, in the south-west for mixed, in central regions of conifer-dominated forest, and in the north-west of the tundra. There is a noticeable lack of fires in the southern-central and north-eastern areas of the study area (Figure 4). Figure 5 shows fire frequency relative to forested area of the cover type. The difference between mean annual frequencies due to cover type is significant (ANOVA: $F = 11.647$, $df = 3, 68$, $p < 0.001$), seemingly because mean annual frequency in broadleaf is one-two orders of magnitude greater than needleleaf and tundra, respectively (Table 2).

Cover Type	Mean Annual Fire Frequency (+- standard error)
Broadleaf	45.59 (+- 12.09)
Mixed	11.71a (+- 2.43)
Needleleaf	1.42a (+- 0.37)
Tundra	0.86a (+- 0.44)

Table 2. Mean annual fire frequency per million ha of forested land, by cover type. Means with same letter are not significantly different (Tukey Multiple Comparison, $p>0.05$).

Temporally, there is no clear trend of fire frequency per million hectares increasing or decreasing over the study period, and there is no significant effect of year on fire frequency when considered across all cover types ($p>0.05$) (Figure 6). Within cover types, however, timing over the study period does seem to have a significant effect on mean fire frequency (ANOVA: $F = 1.49$, $df = 51, 17615$, $p < 0.05$). Tukey Multiple Comparison reveals significant differences occur in fire frequency in: 2009 and 2010 for broadleaf, 2002 for mixed, 2011 for needleleaf and 2018 for tundra compared to all other years for each cover type. The latter two can be seen from the MODIS maps – those in the central portion of the needleleaf zone mostly occur in 2011 (in yellow, Figure 4); while the cluster in the north-west of the tundra-forest boundary predominantly occur in 2018 (in purple, Figure 4). It is particularly striking that peaks in frequency are generally followed by troughs of little to no fire activity (Figures 5, 6).

The total number of fires by month also shows a clear difference in when fires occur, both spatially and across the study period (Figure 6). Fires in mixed forest and the tundra are consistent throughout the study duration – occurring, respectively, much earlier (March-April) and later (July and August) in the season. In contrast, fire frequency is distributed much more evenly across the season in coniferous-dominated forest during the study period. For broadleaf, there appears to be a shift in fire frequency from predominantly mid-late season to early-mid season from 2009 (Figure 6).

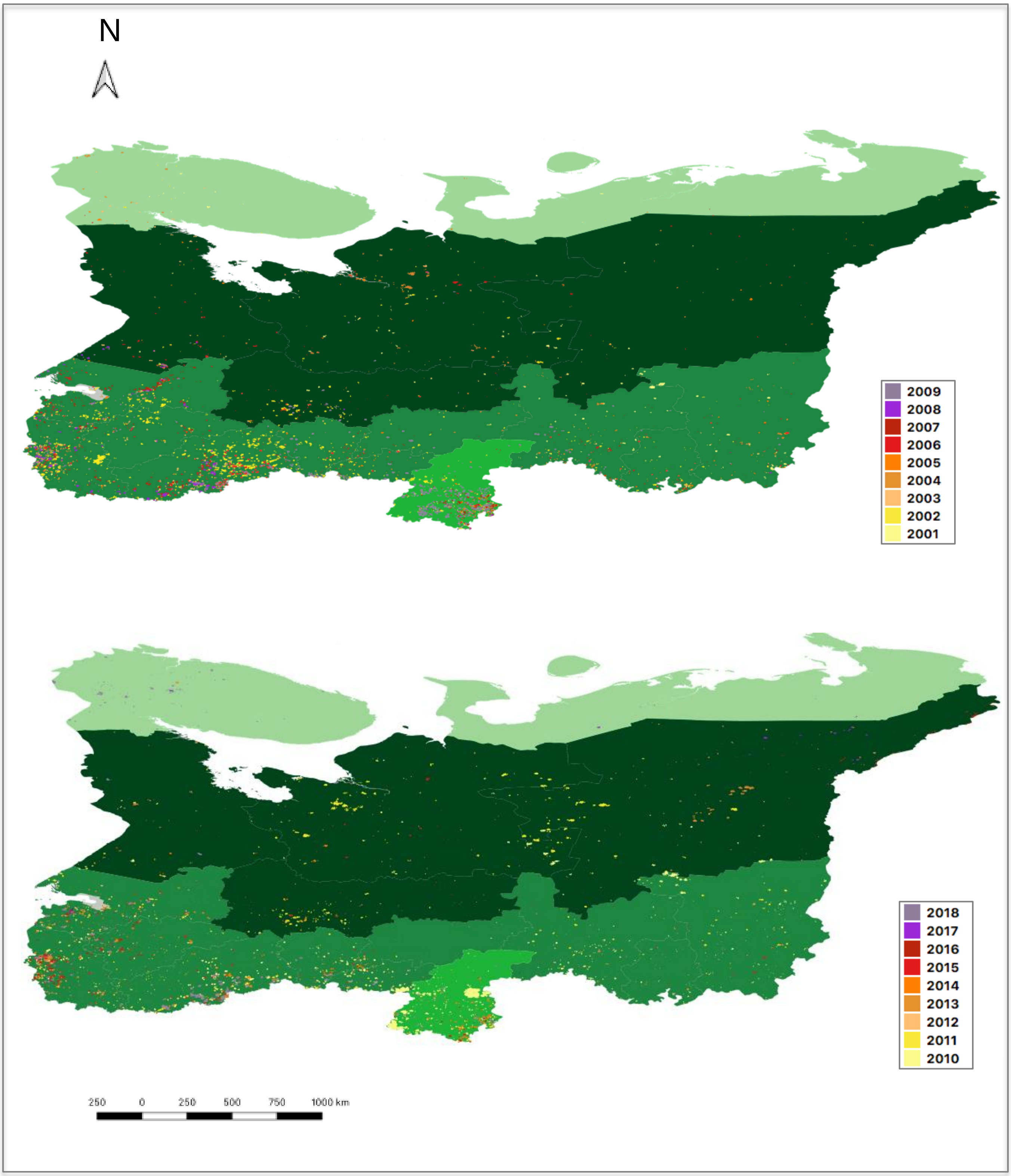


Figure 4. Fires recorded by MODIS burned area product over the study area, verified with LandSat data in certain locations, 2001-2009 (top) and 2010-2018 (bottom).

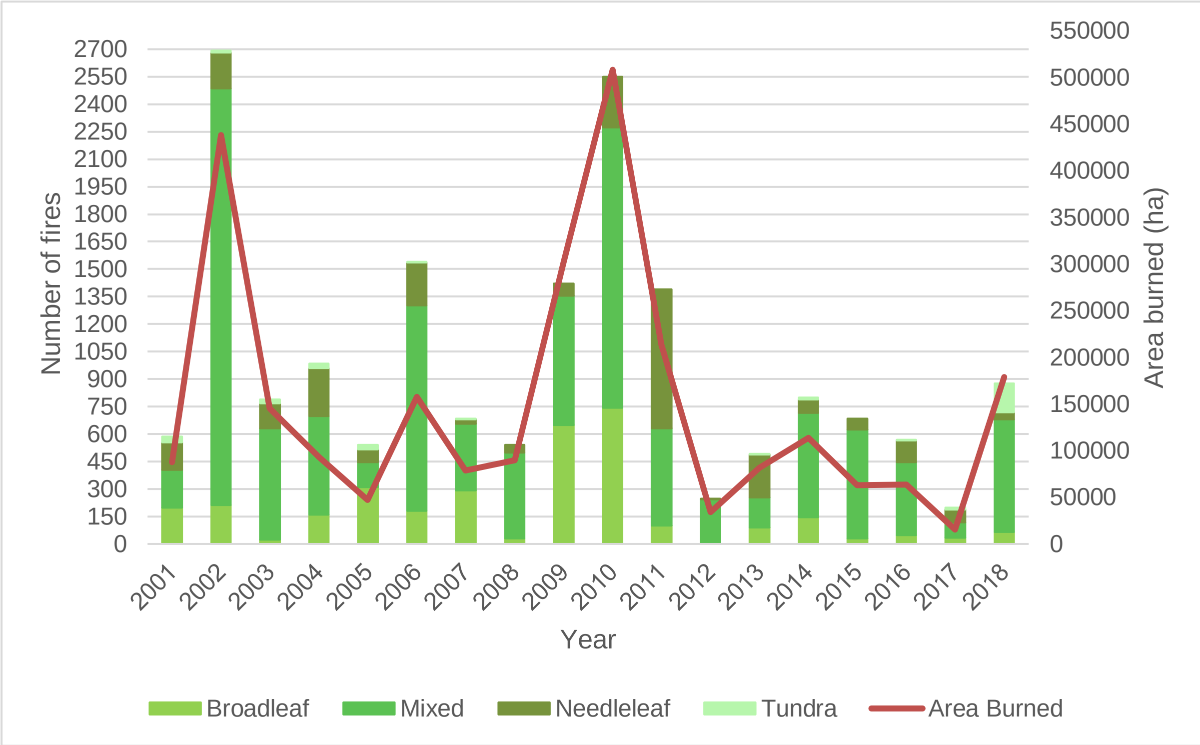


Figure 5. Total number of fires by cover type and total area burned over the study period.

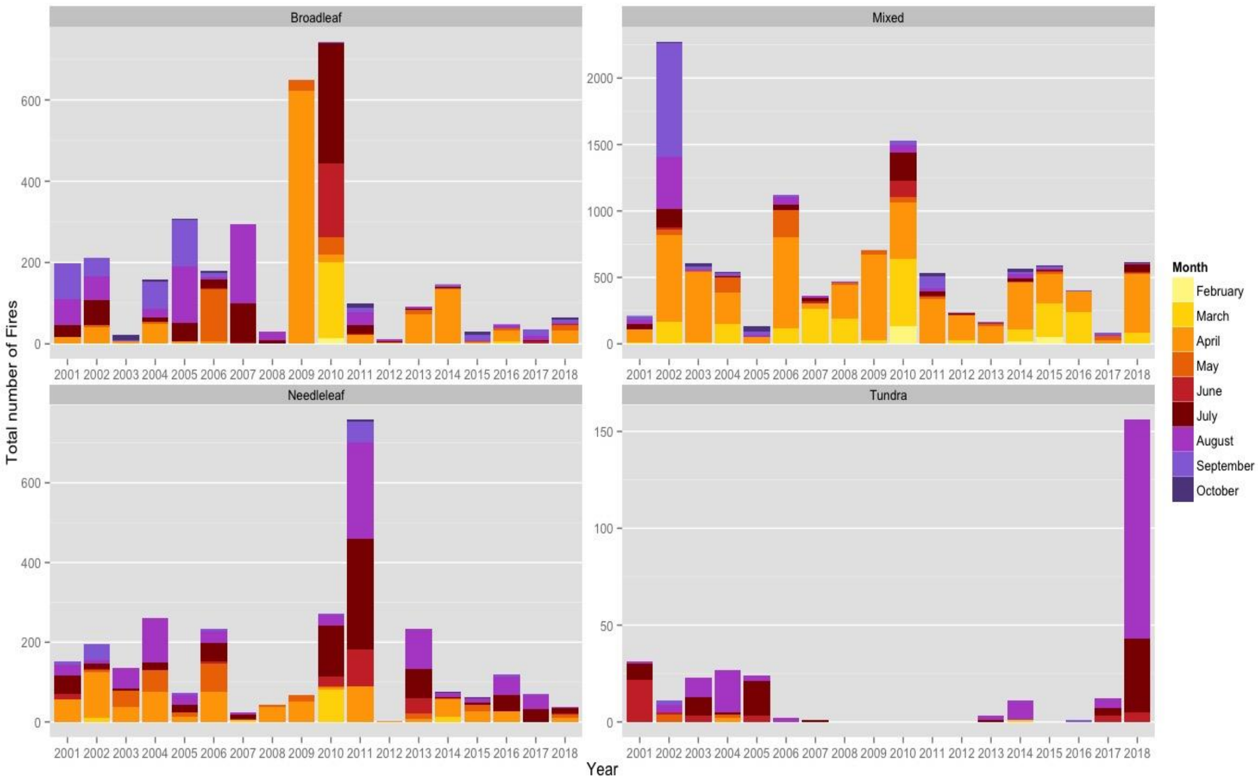


Figure 6. Distribution of fires throughout the fire season for each cover type.

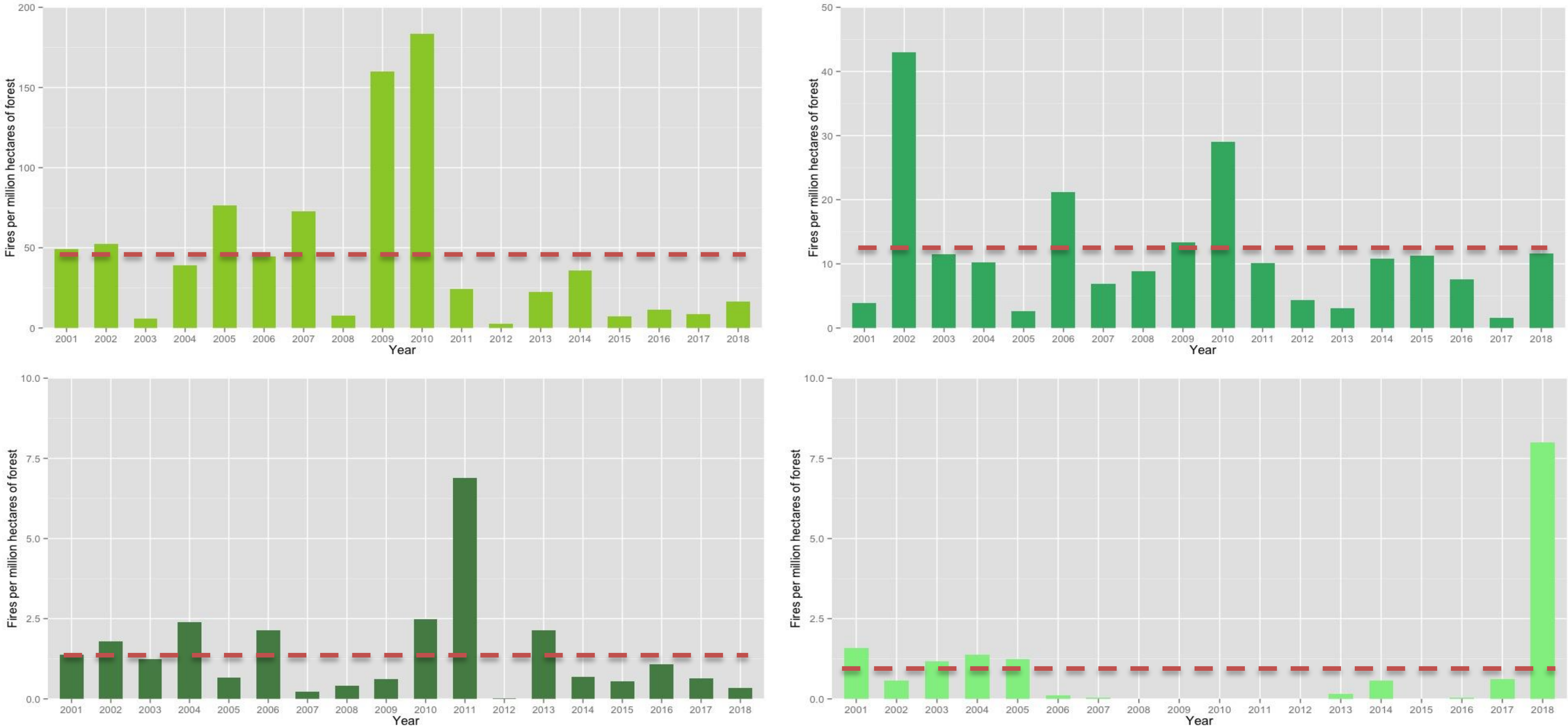


Figure 7. Annual number of fires per million hectares of forested land for each of the land cover types, with overall mean annual fire frequency per million hectares of forested land represented by red-dashed line. Clockwise from top left: Broadleaf, Mixed, Needleleaf, Tundra. Note the different scales between the plots, given significant variation in frequency between cover types.

ii. Intensity

In total, just over 2.7 million hectares of land, or approximately 1.4% of forested land, was affected by fire across the study region. Figure 8 shows the mean size of fires by cover type and year. Ranging from around 50-350 hectares, fires across the study area are of variable size. There is a mean size across the area and whole study period of approximately 154 hectares. There is no significant variation in annual average fire size according to year ($p>0.05$). While ANOVA statistical analysis reports a significant spatial difference between annual average size by cover type (ANOVA: $F = 3.473$, $df = 3, 17615$, $p < 0.05$), Tukey multiple comparison test presents no significant difference between means (all p-values of comparisons > 0.05). There is, however, a significant difference on mean annual fire size by cover type, averaged across the *whole* study period (ANOVA: $F= 3.627$, $df = 3, 17615$, $p<0.05$). This is generated by larger fires in broadleaf compared to needleleaf and tundra (Figure 6, Table 3).

Cover Type	Mean Annual Fire Size (ha) (average fire size per year)
Broadleaf	185.54a
Mixed	151.21ab
Needleleaf	135.99b
Tundra	81.45b

Table 3. Mean annual fire size by cover type. Means with the same letter are not significantly different ($p>0.05$, Tukey Multiple Comparison).

iii. Severity

Values of delta Normalised Burn Ratio (dNBR) were calculated for each location up to four years post-burn (Figure 9). Immediate post-burn values range from low-

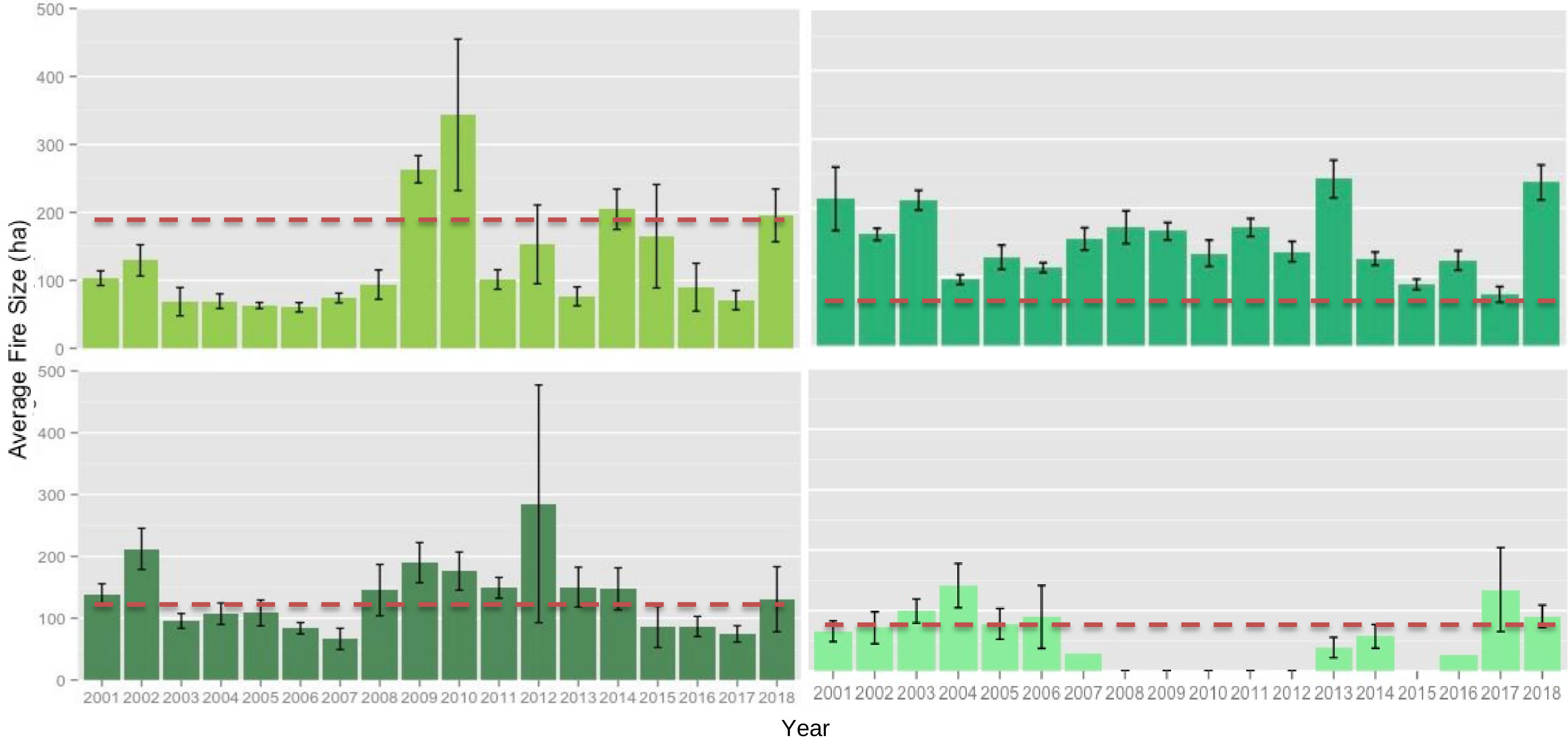


Figure 8. Mean fire size by year and cover type (+/- standard error), with red-dashed line representing mean annual fire size for the cover type over the study period. Clockwise from top left: Broadleaf, Mixed, Needleleaf, Tundra.

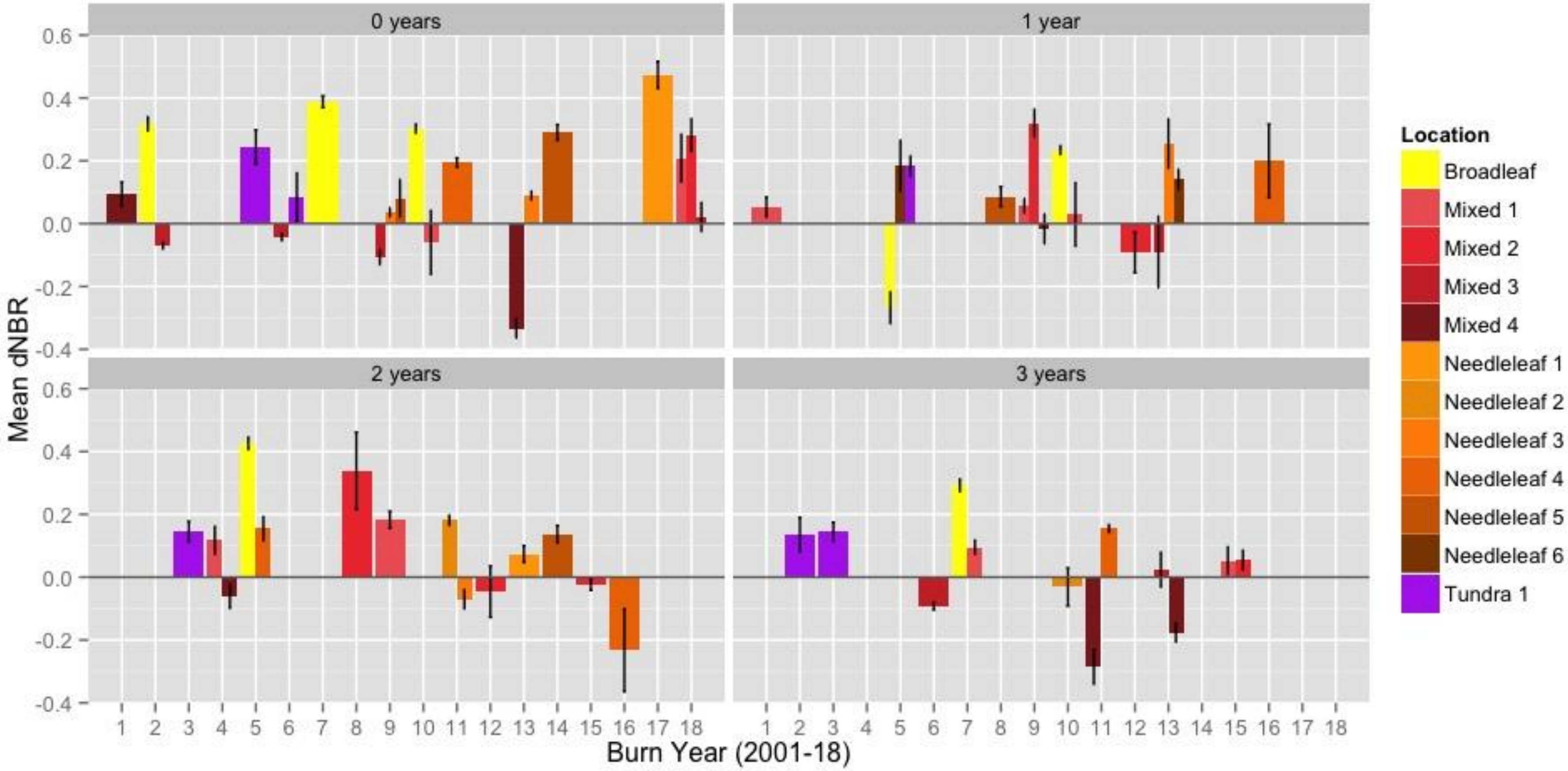


Figure 9. Mean dNBR value 0-4 years post-burn for fires in each location, derived from LandSat imagery. Note some locations do not have imagery across all years, and there was no imagery available within 3 years of burns in Tundra 2 (which mainly occurred in 2018).

moderate severity in the tundra and some needleleaf locations (around 0.1-0.3), to moderate-high severity in broadleaf and needleleaf in later burn years (around 0.35-0.6). Generally, values decrease as time passes from the initial burn, but few go below zero and the rate of change appears to vary quite significantly between years and locations. Consider for example the one-year post burn value for broadleaf compared to one-year post burn values in needleleaf and the tundra (Figure 9). Interestingly, mixed forest values zero-years post burn are negative (Figure 9). Overall, there was no significant effect of burn year on altering dNBR values zero and one years after burning in any of the mixed forest locations or within the tundra (of which only one location had imagery 0-1 years post-burn) ($p > 0.05$). There was a significant effect of burn year on altering dNBR values zero years after burning within needleleaf-dominated forest, however (ANOVA: d.f. = 8, 654, $F = 12.2559$, $p < 0.01$). These differences were generated by more severe dNBR values in 2011, 2014 and 2017 compared to 2009. In broadleaf, dNBR values of burns in 2007 zero-years after burning were significantly more severe than in 2010, while 1-year post burn values were still more severe for burns in 2010 compared to 2005 (ANOVA: d.f = 3, 998, $F = 9.17$, $p < 0.01$).

2. Natural and Anthropogenic Controls

The following images (Figures 10-17) show fire occurrence against the ECMWF temperature anomalies recorded for the study area between 2001 and 2018. There is some agreement where fire occurrence matches the spatial variation in temperature anomalies, particularly in the summer months of 2004, 2010, 2011 and 2015. However, there are times when lots of fires occur in normal or cooler than average temperatures - mainly the springtime months (April and May) across all years.

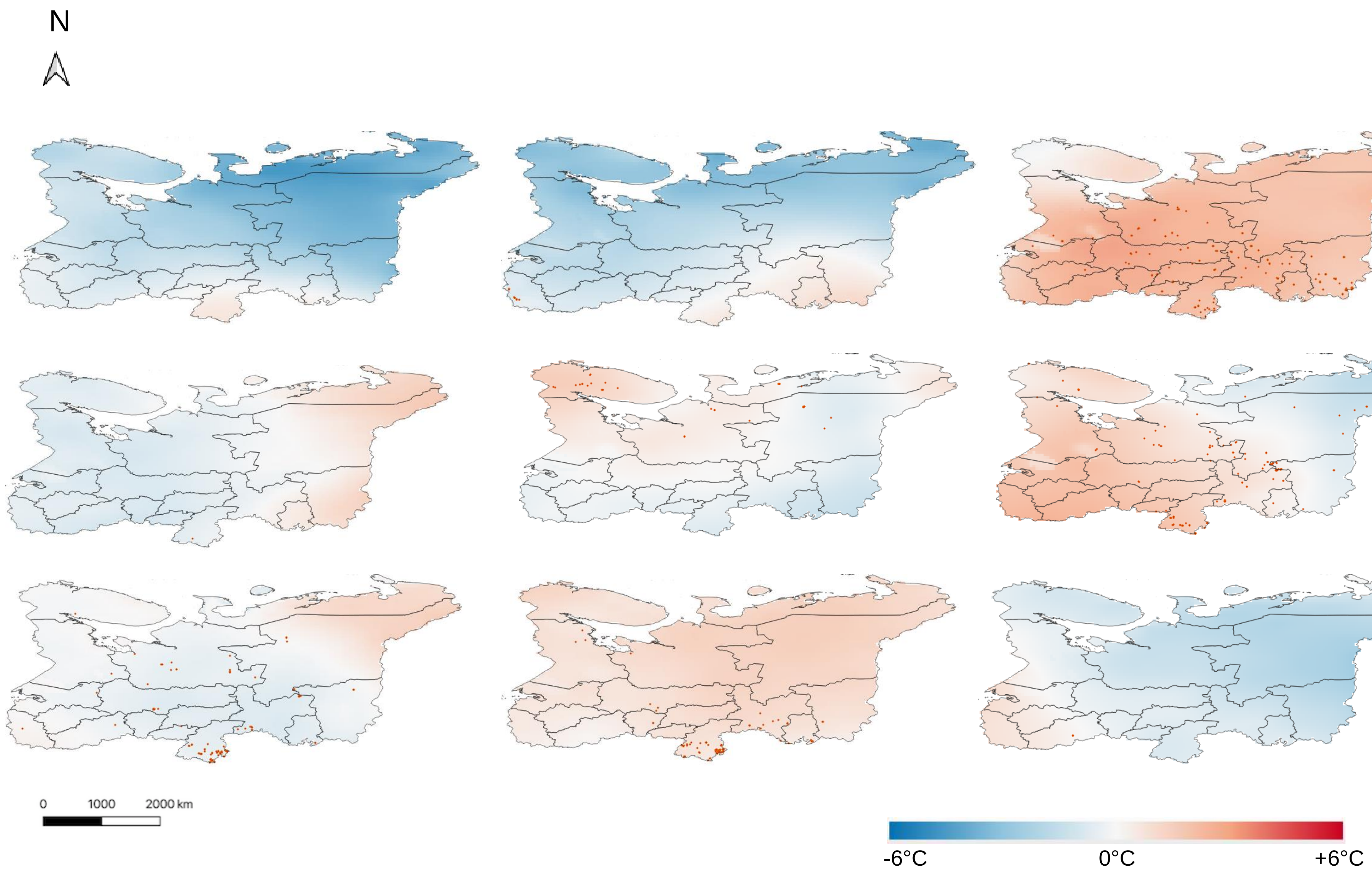


Figure 10: 2001 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

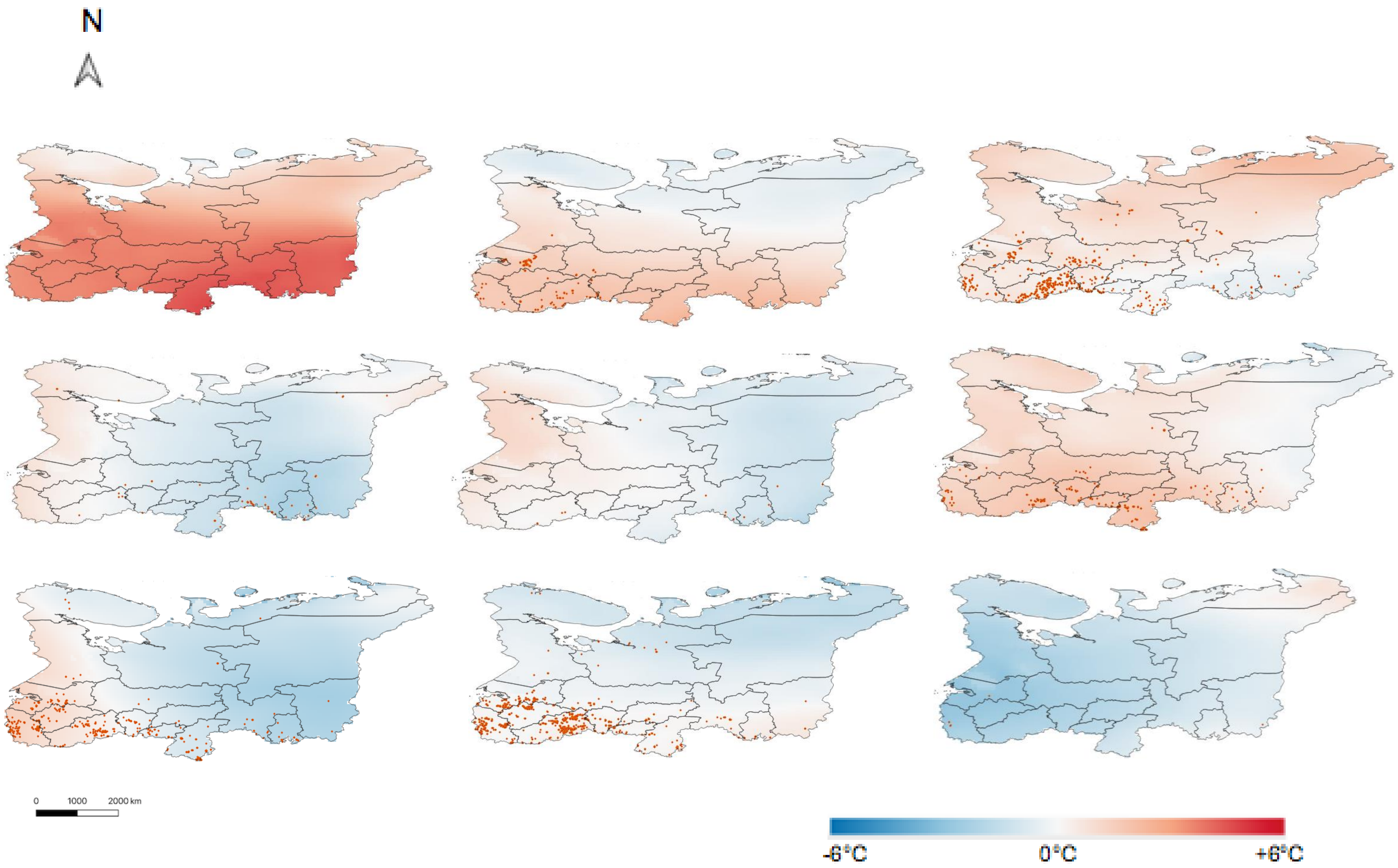


Figure 11: 2002 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

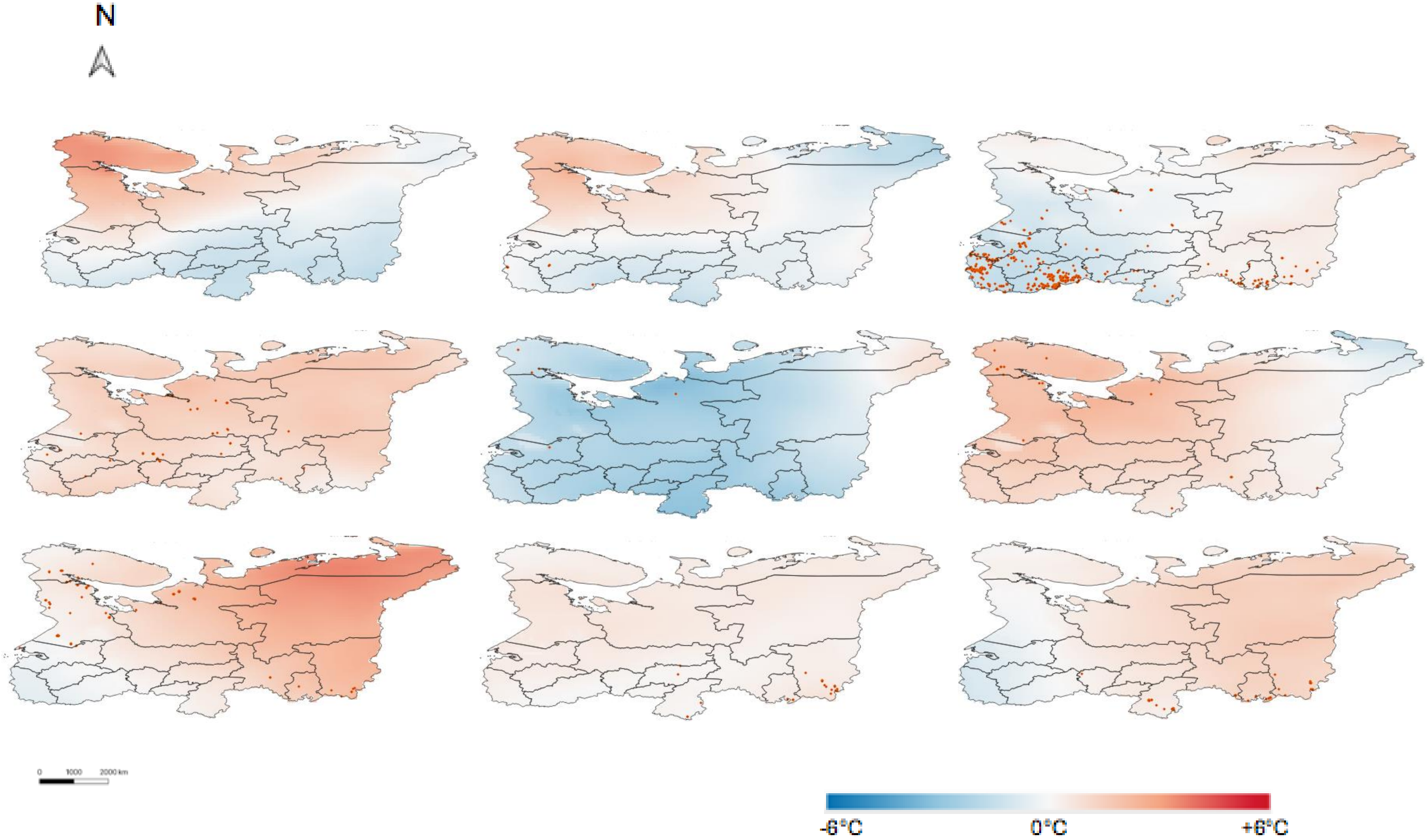


Figure 12: 2003 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

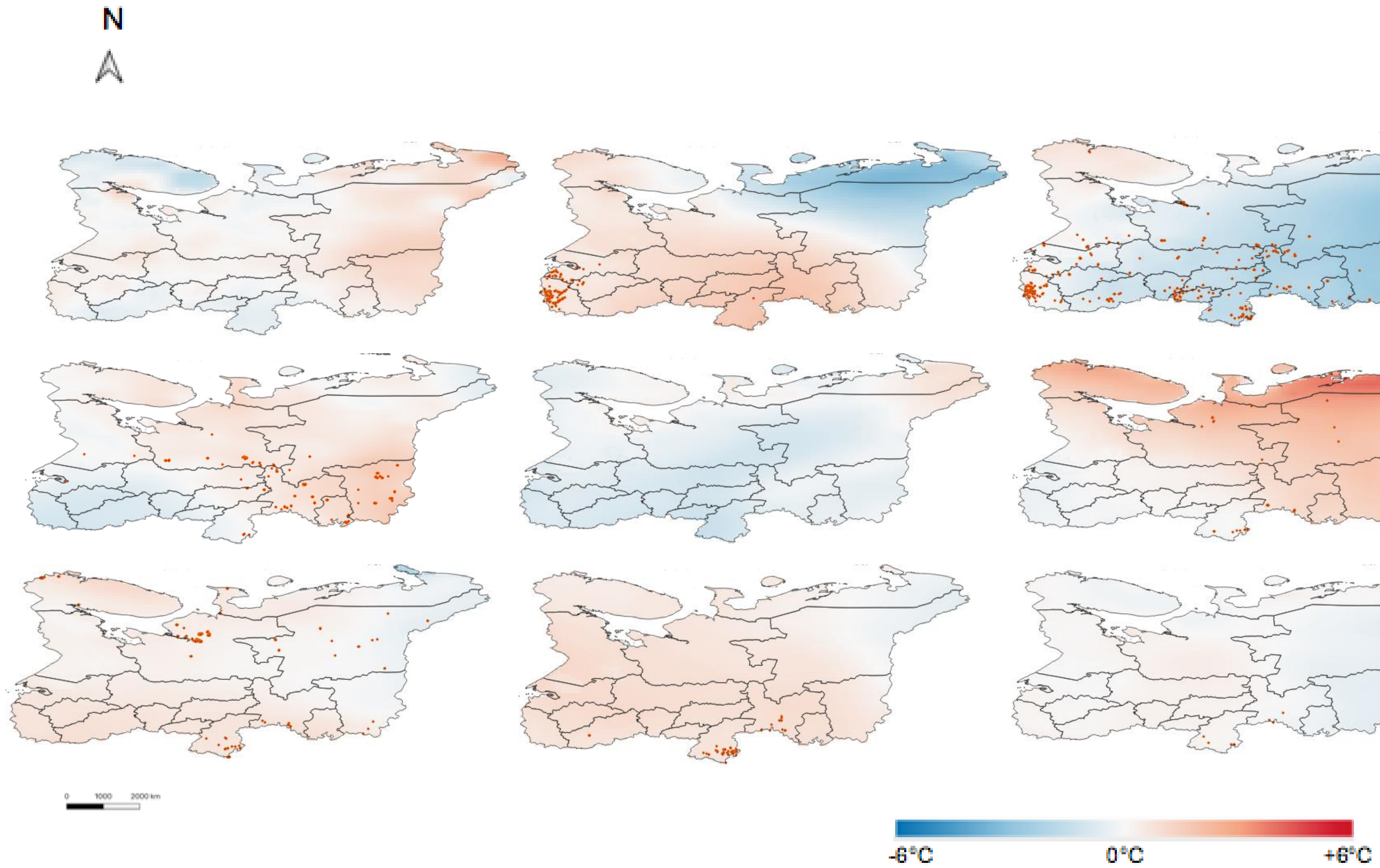


Figure 13: 2004 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

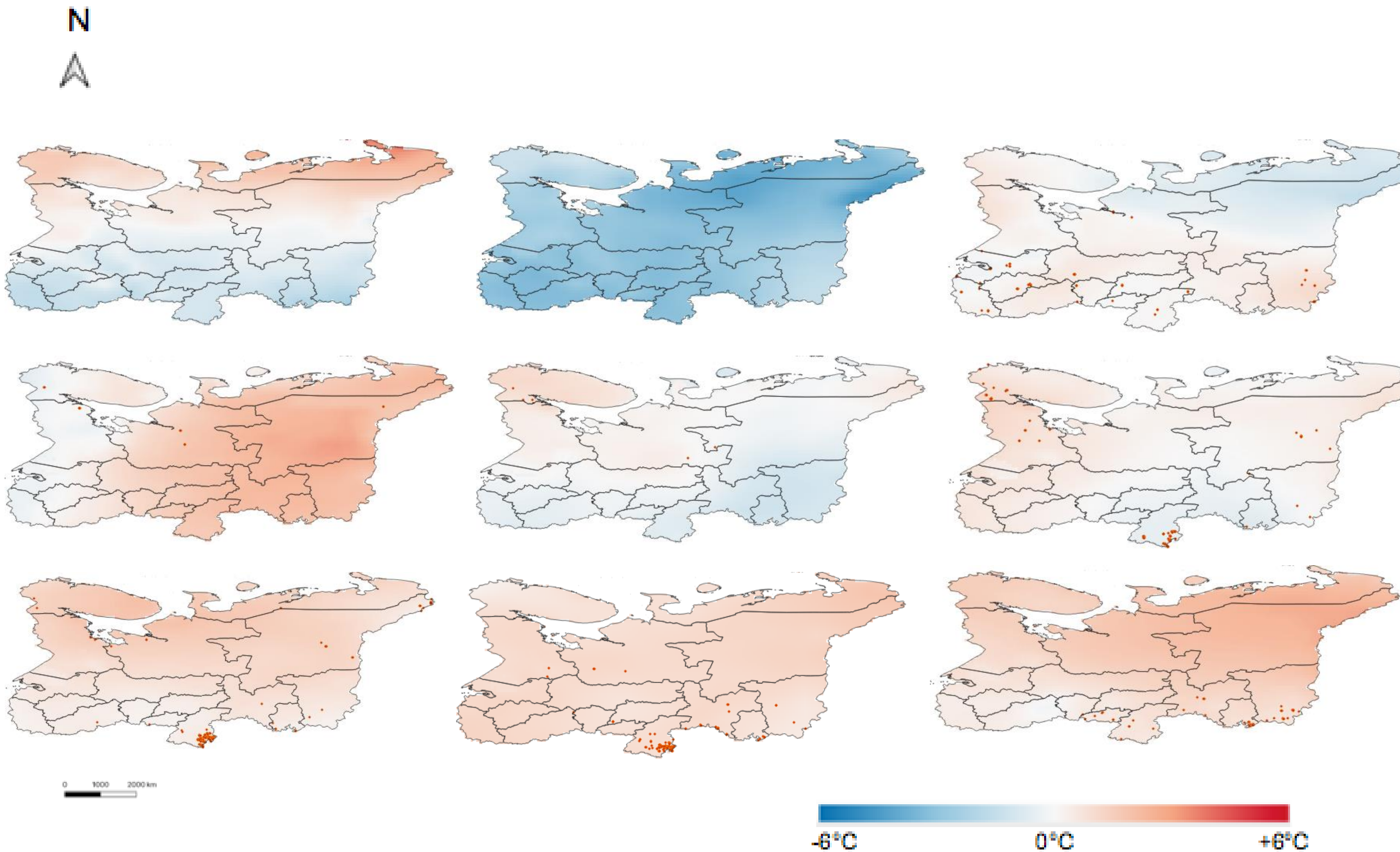


Figure 14: 2005 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

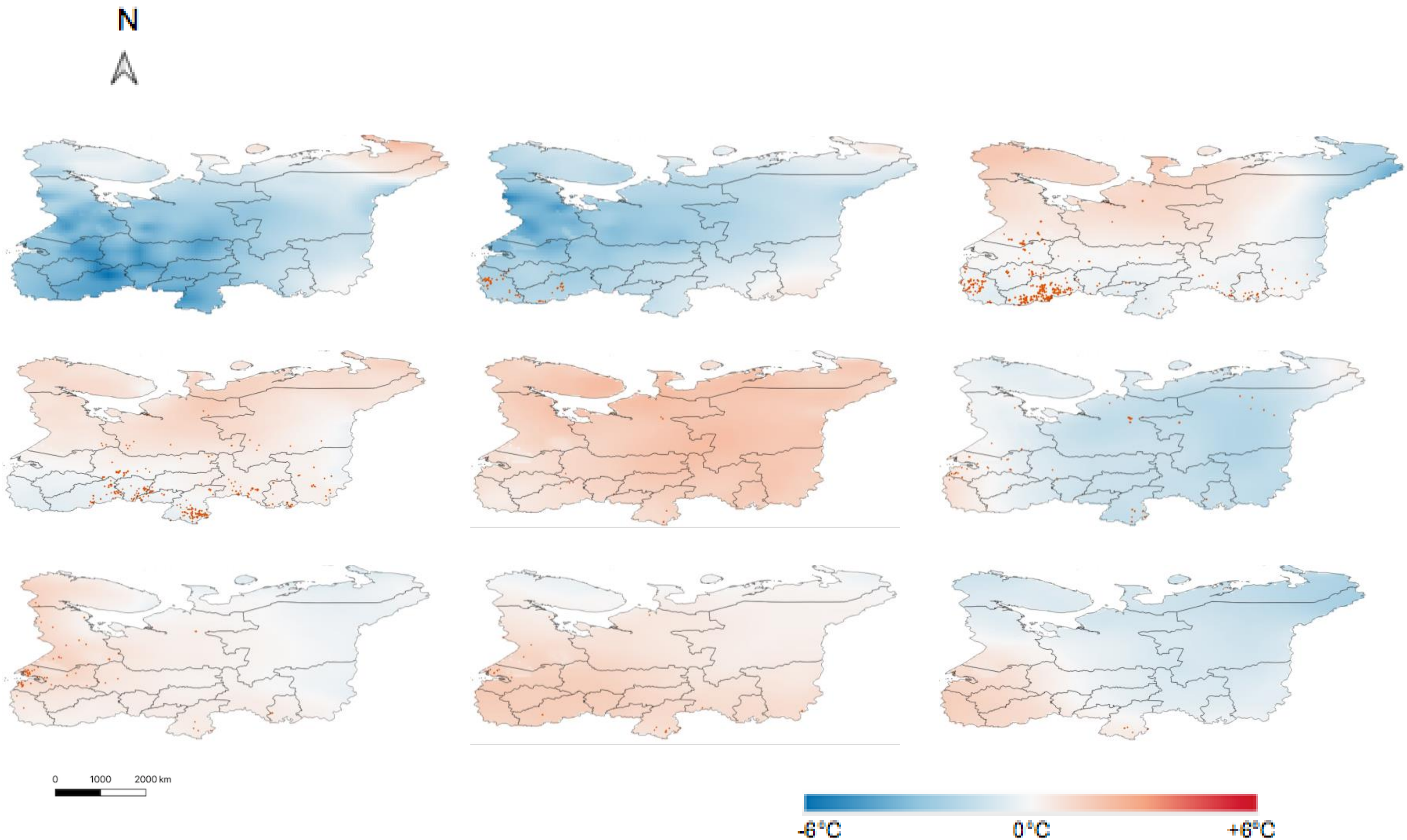


Figure 15: 2006 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

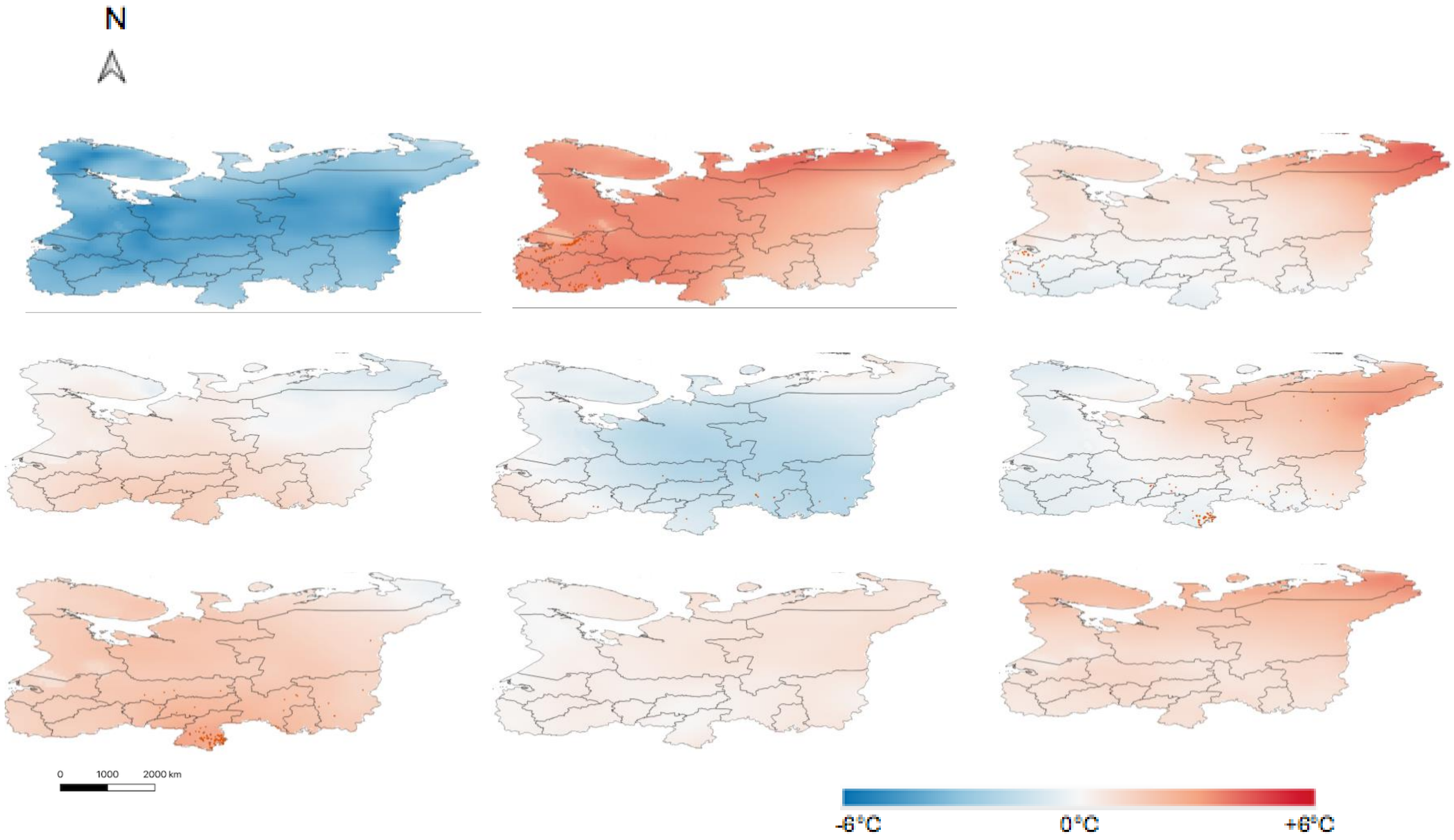


Figure 16: 2007 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

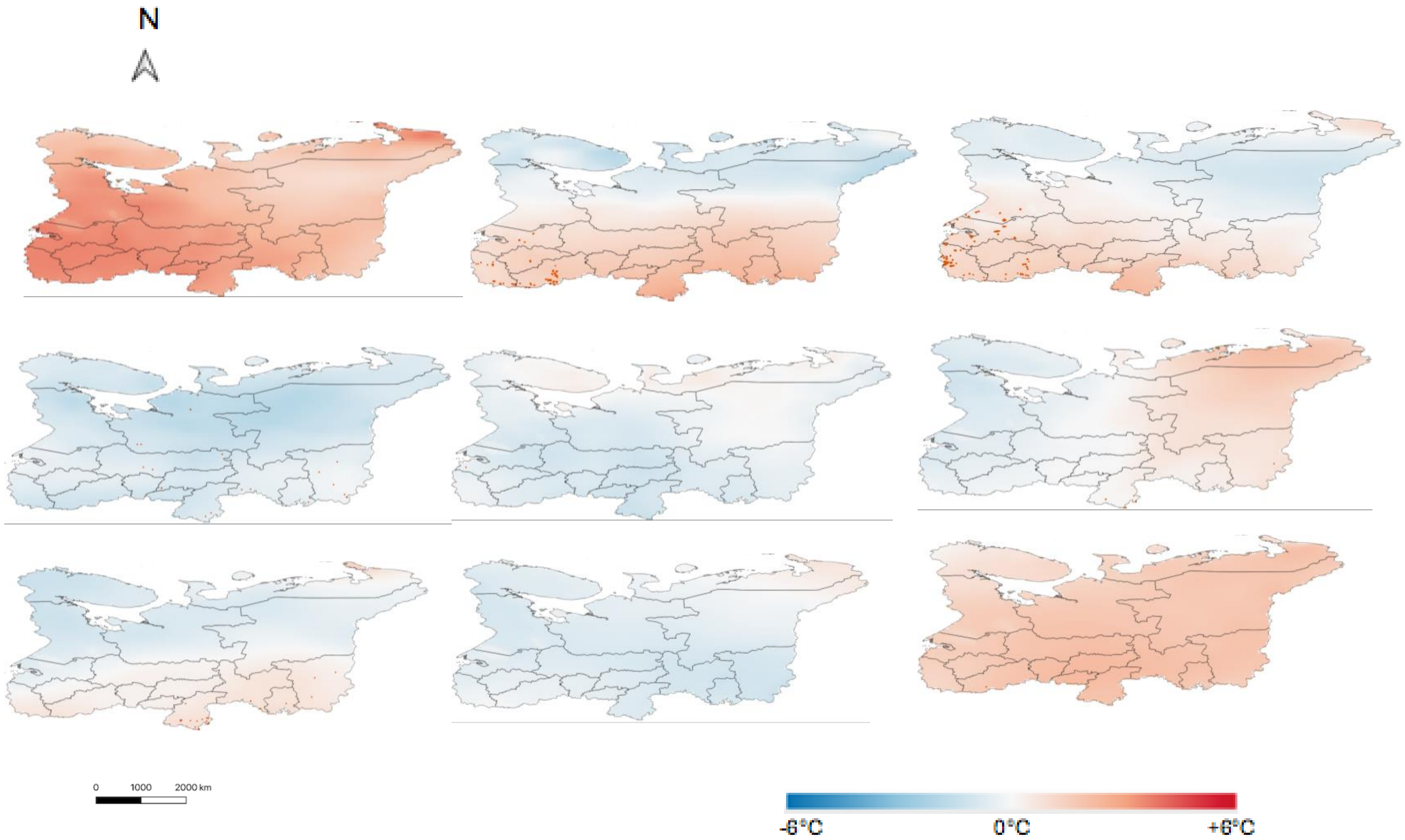


Figure 17: 2008 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

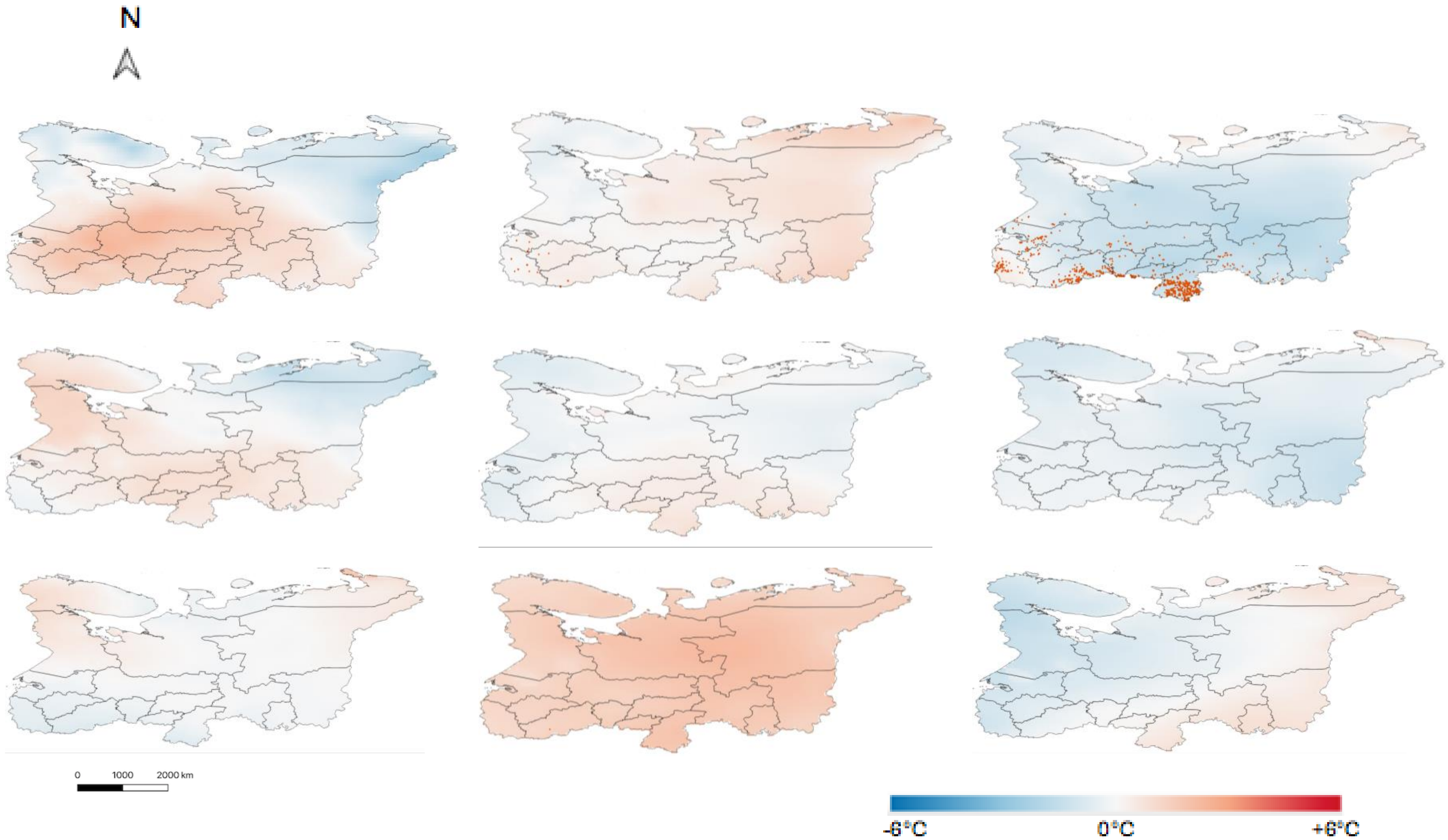


Figure 18: 2009 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

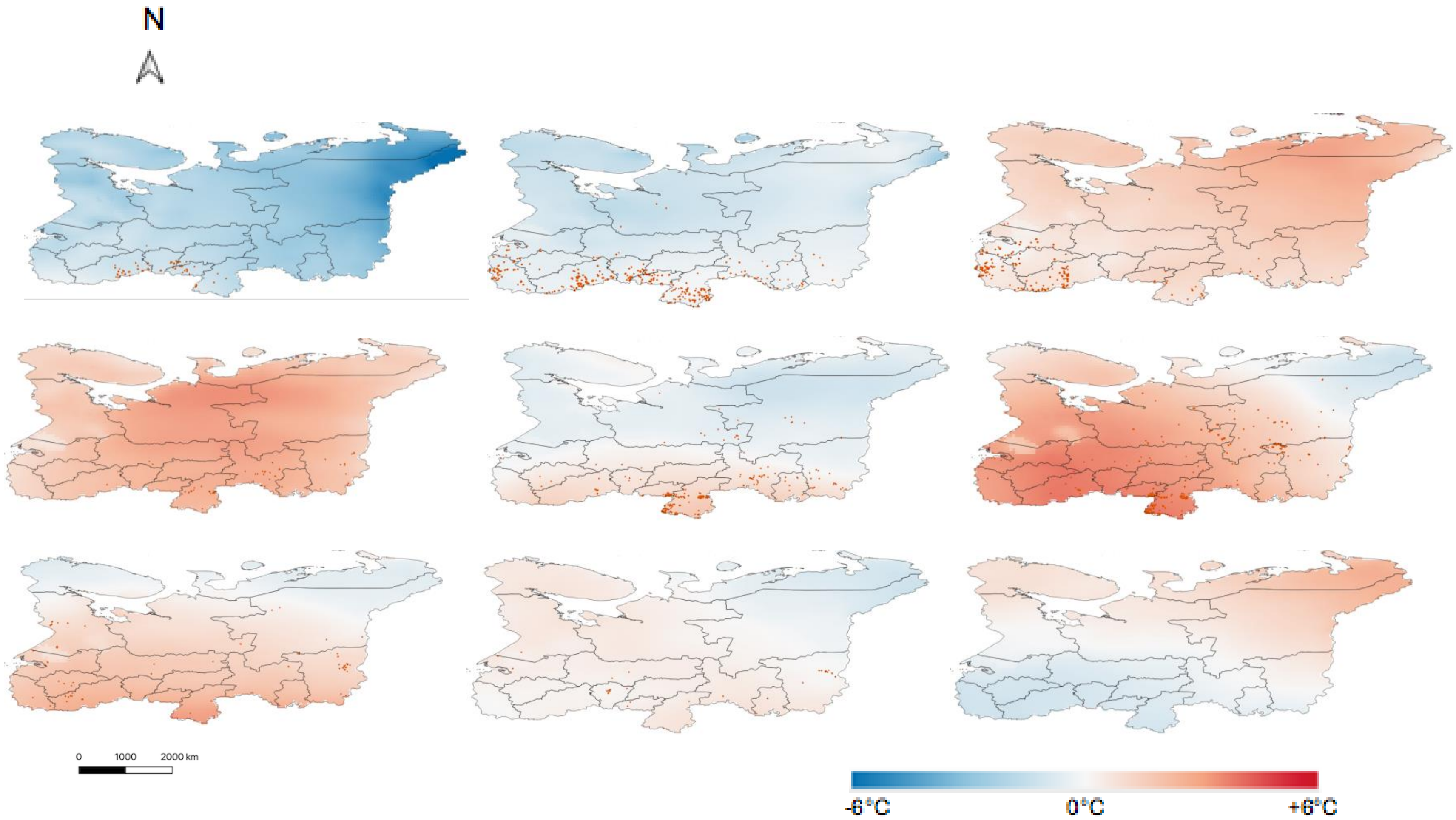


Figure 19: 2010 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

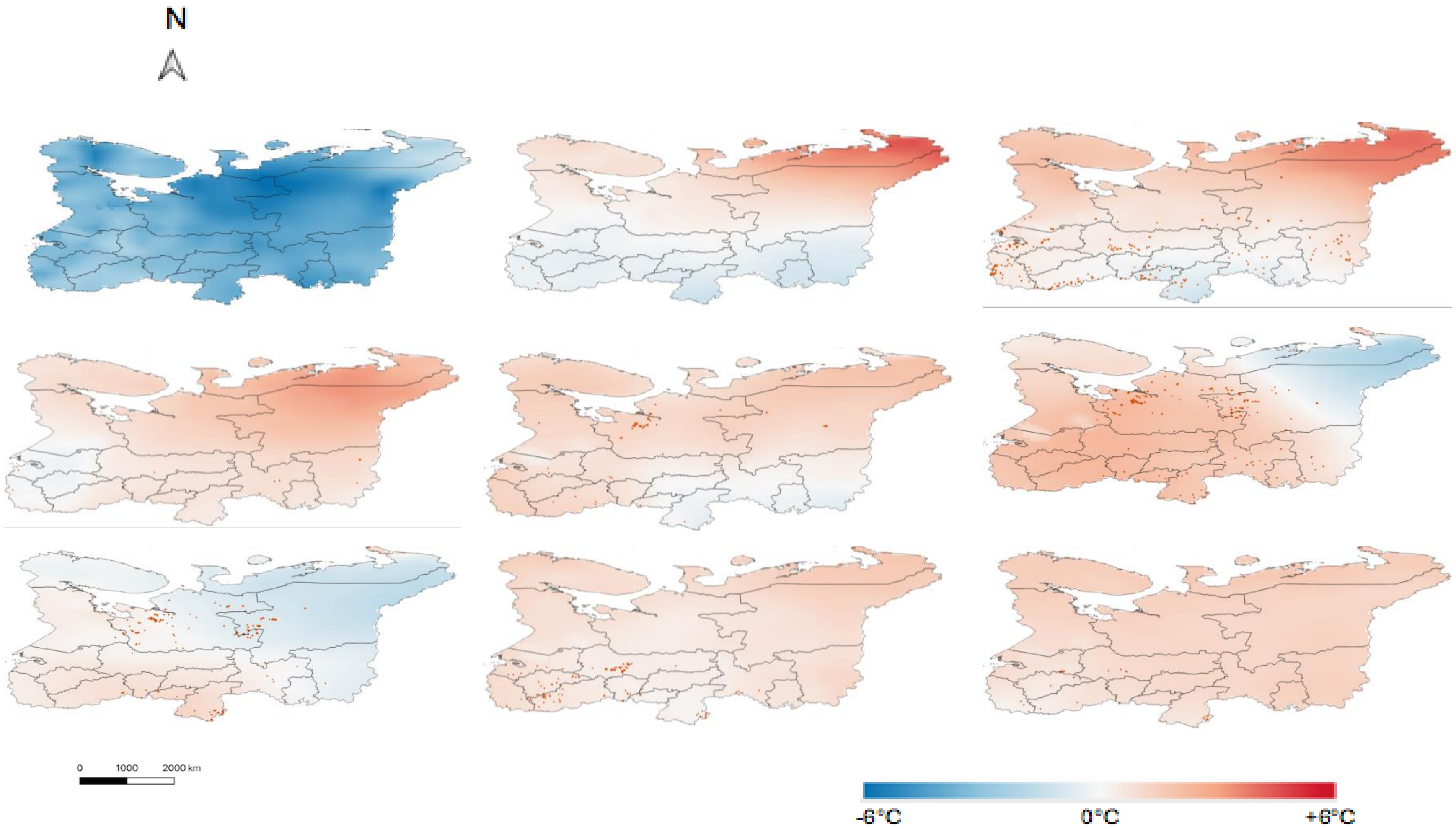


Figure 20: 2011 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

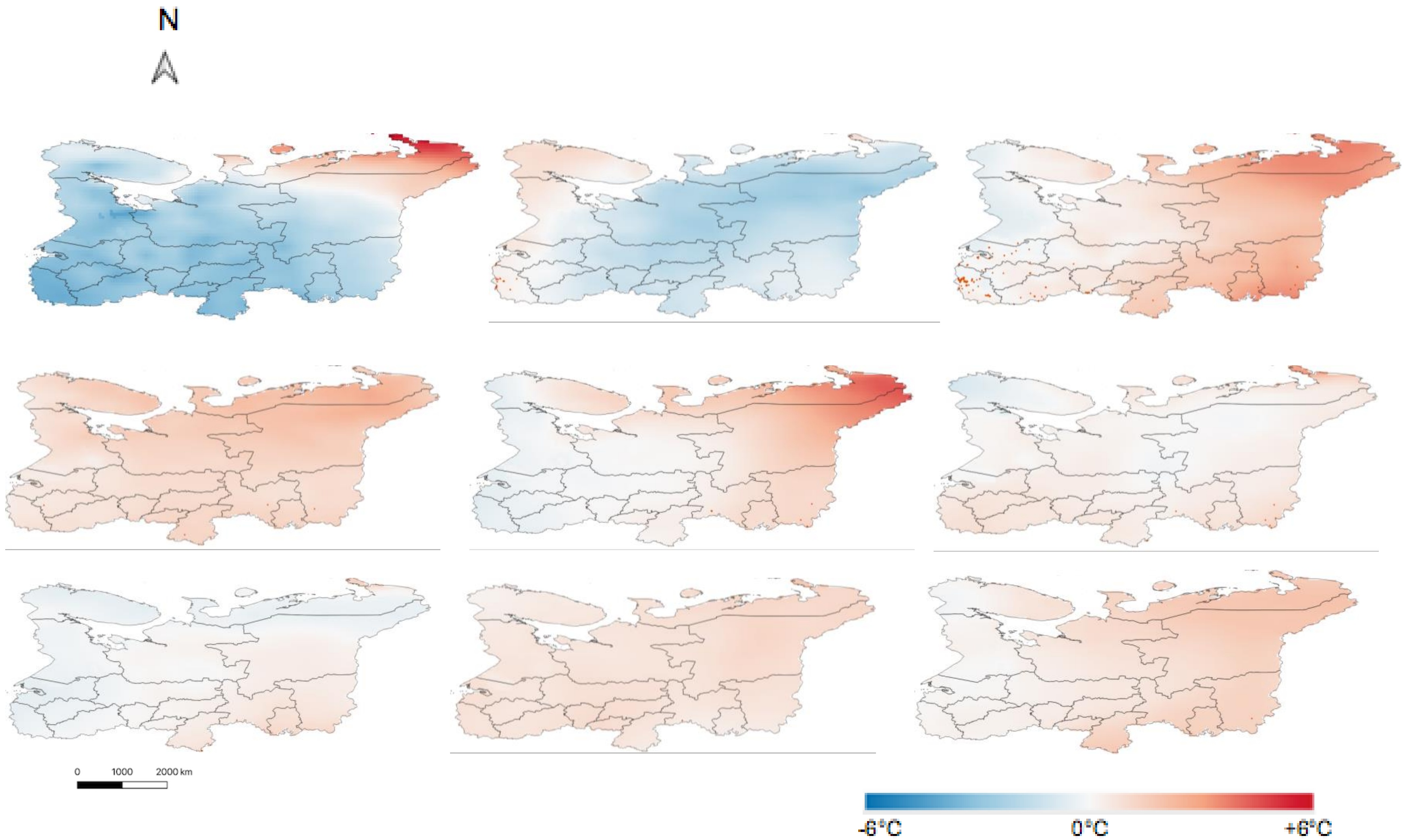


Figure 21: 2012 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

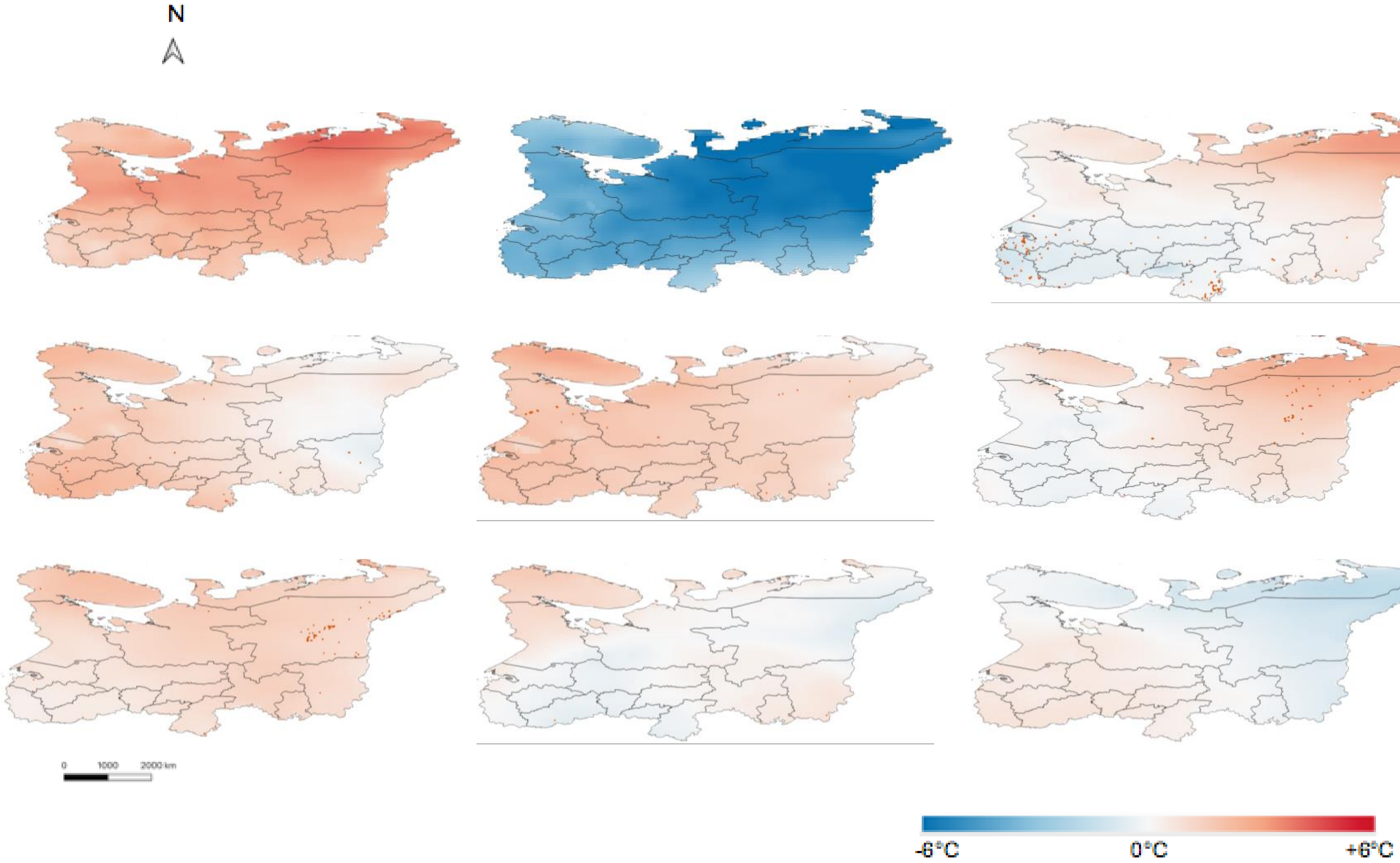


Figure 22: 2013 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

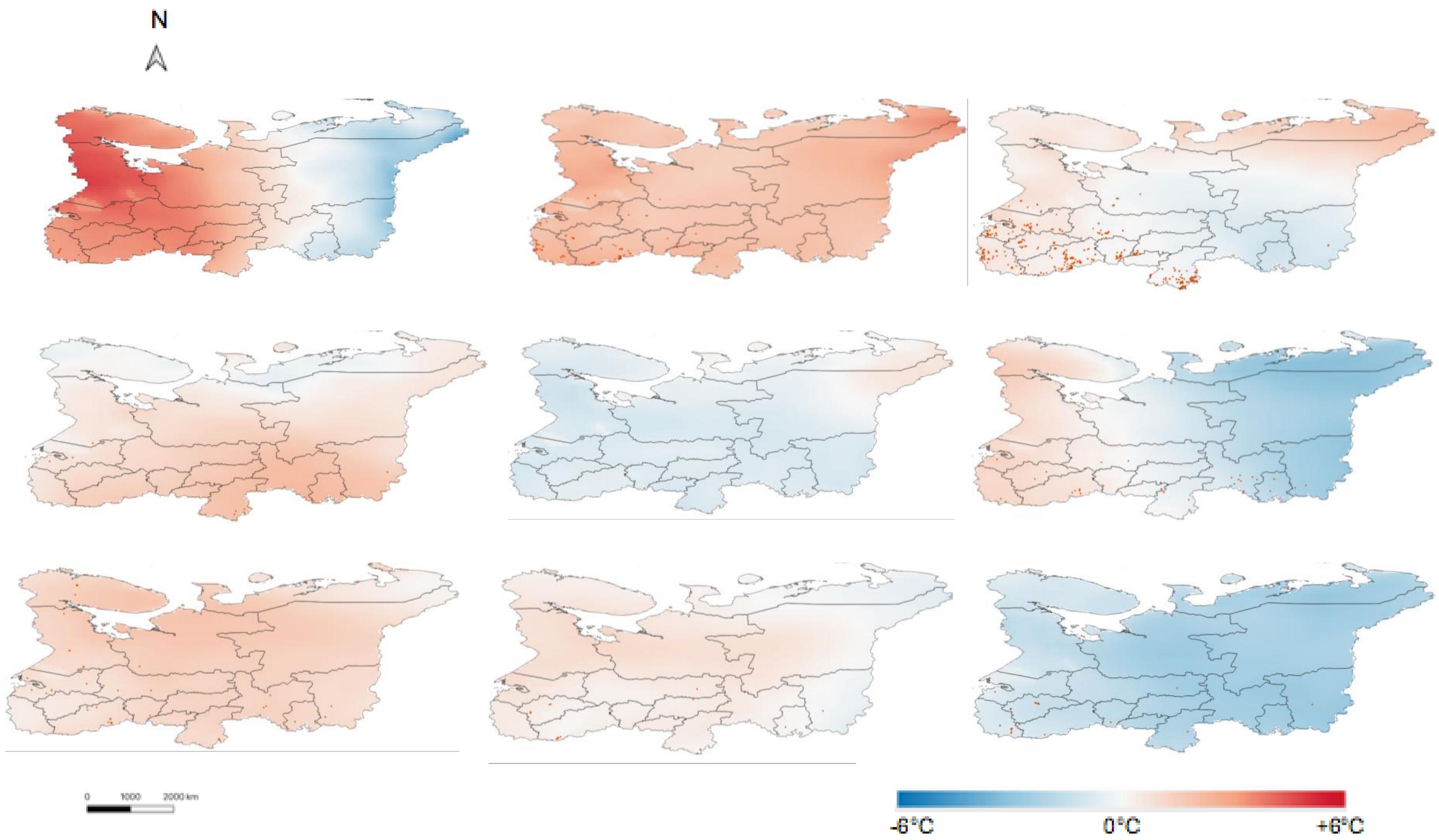


Figure 23: 2014 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

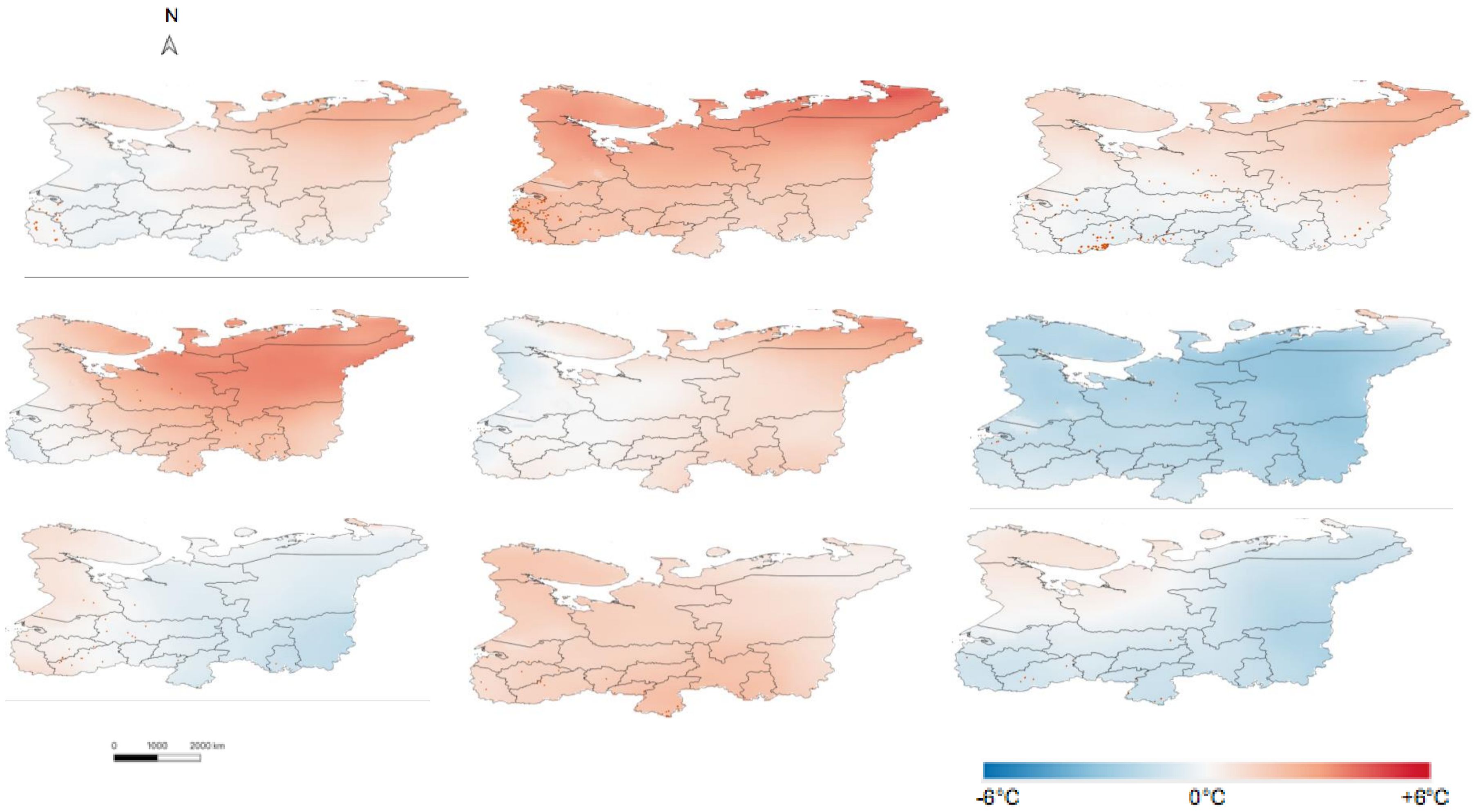


Figure 24: 2015 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

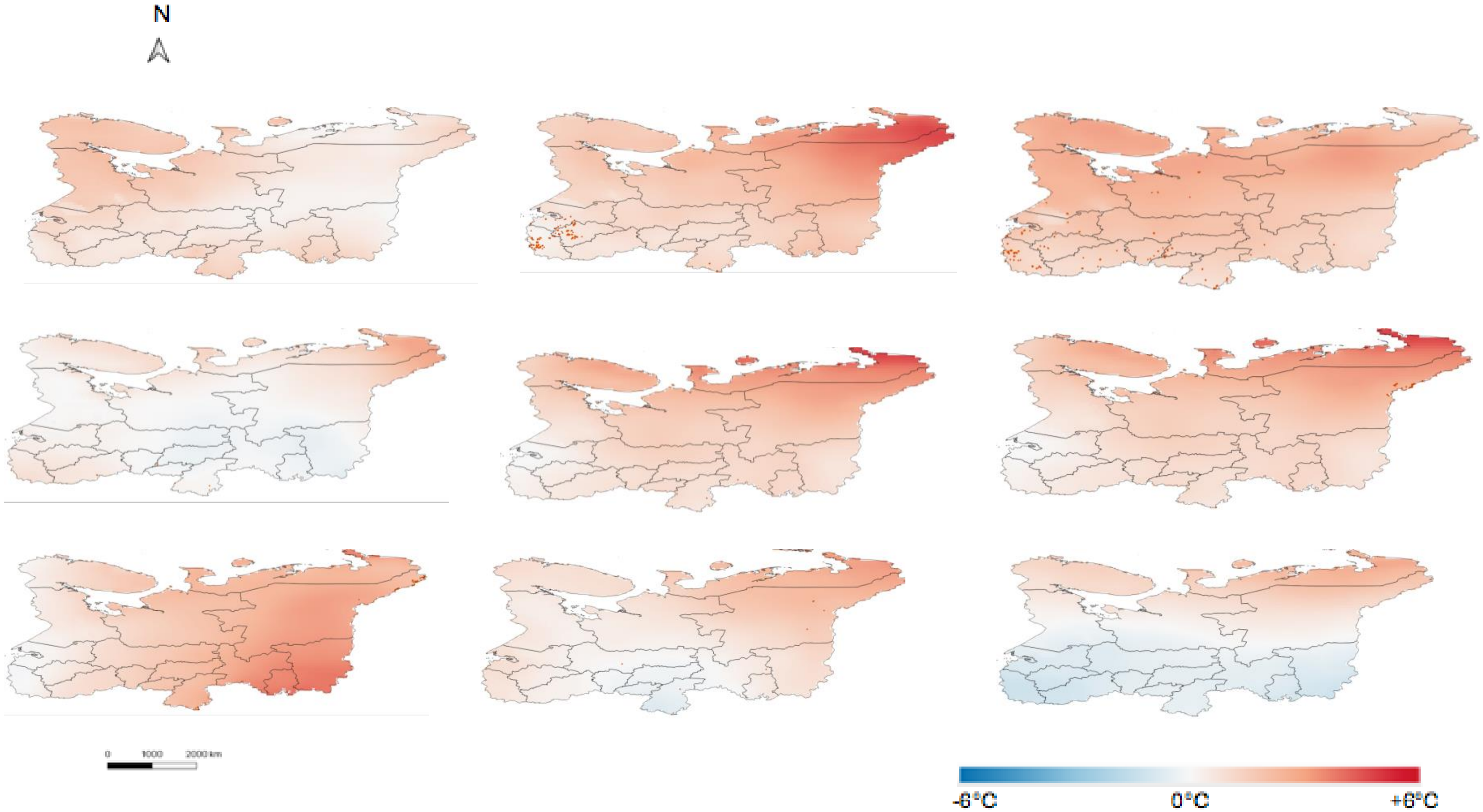


Figure 25: 2016 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

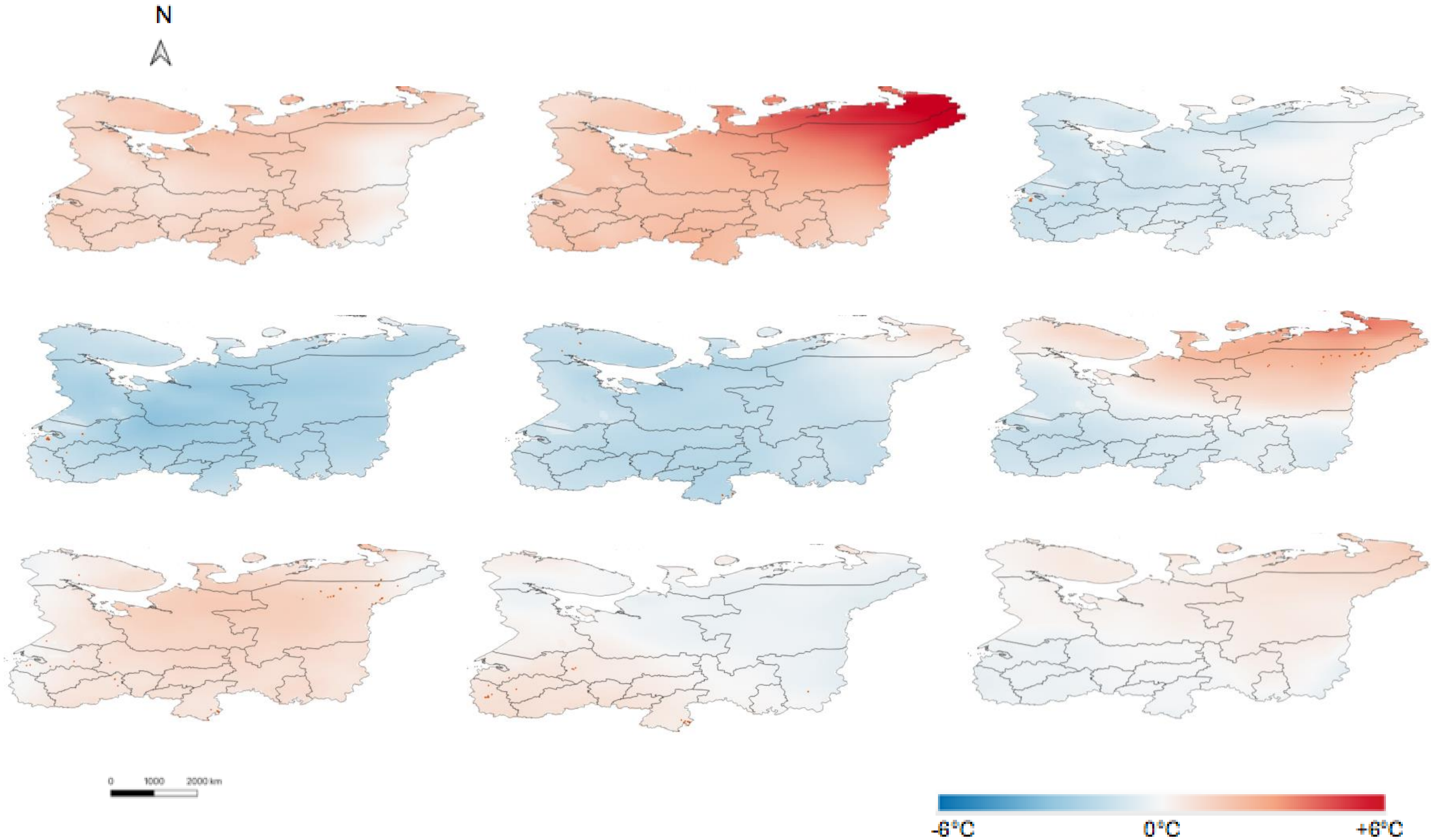


Figure 26: 2017 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

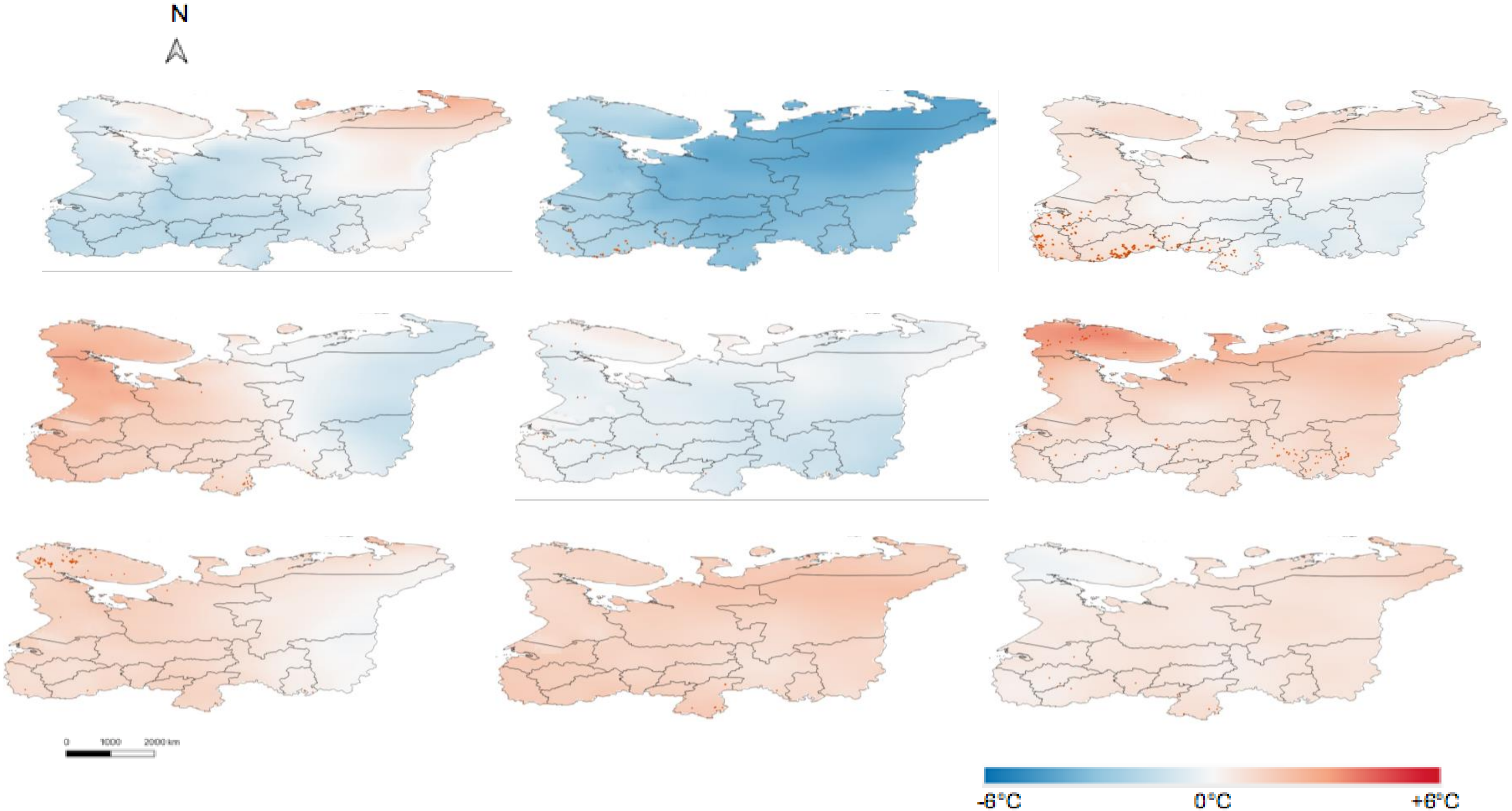


Figure 27: 2018 MODIS burned area and ECMWF monthly surface temperature anomalies during the fire season. Top row (left-right): February, March, April; Middle row: May, June, July; Bottom row: August, September, October.

ii. Anthropogenic

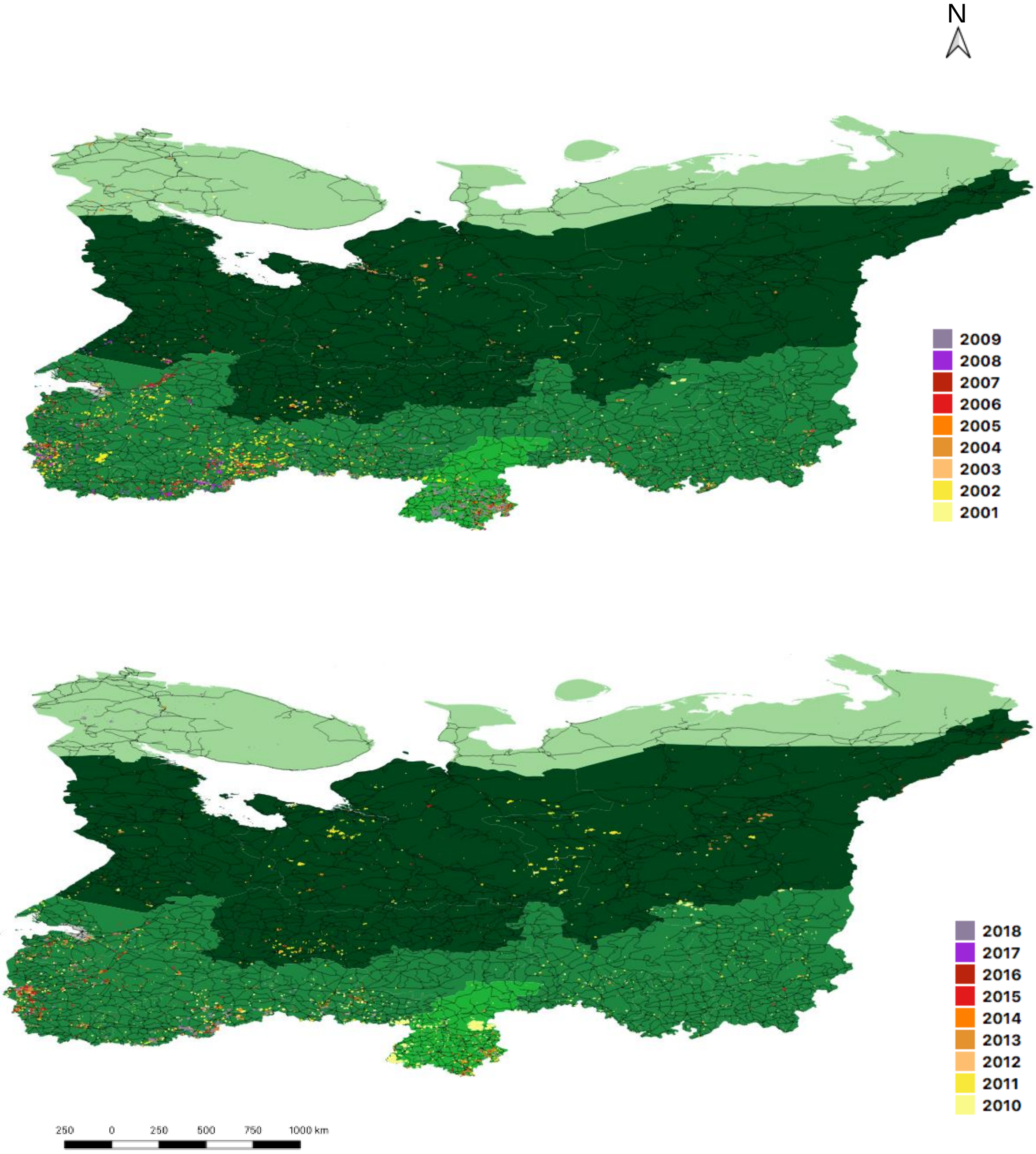


Figure 28. Spatial distribution of fires across the study area in comparison to the road network. Top: 2001-2009, bottom: 2010-2018.

There is a significant negative relationship between the frequency of fires and proximity to roads in broadleaf ($y = 71.39 + -4.45x$, $F = 30.97$, d.f. =1, 286, $p<0.01$) and in needleleaf ($y = 7.44 + -0.16x$, $F = 241.28$, d.f. = 1, 1078, $p<0.01$) (Figure 28). Road proximity accounts for 9% and 18% of variation in fire frequency for each of these locations respectively (Multiple R^2 squared statistic). There is no significant relationship between road proximity and the frequency of fires in tundra or mixed forest. There is a significant positive relationship between the size of fires and proximity to roads in broadleaf ($y = 49.40 + 19.96x$, $F = 4.8$, d.f = 1, 184, $p < 0.05$) and mixed forest ($y = 112.59 + 7.065x$, $F = 7.64$, d.f. = 1, 314, $p<0.01$) (Figure 29). Road proximity accounts for just 2% of the variation in fire size for each of these locations (Multiple R^2 squared statistic). There is no significant relationship between fire size and road proximity in tundra or needleleaf.

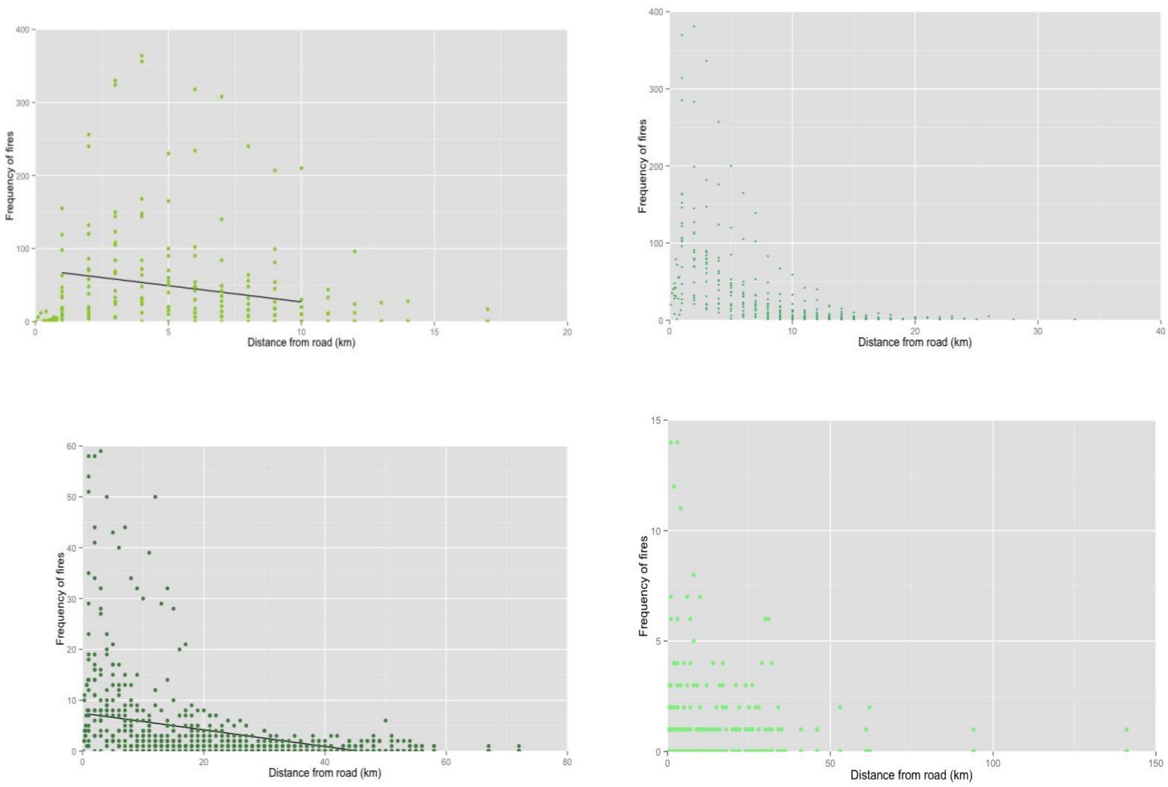


Figure 29. The relationship between fire frequency and road proximity. Relationships with a regression line were significant (see paper for equation). Clockwise from top left: broadleaf, mixed, needleleaf and tundra. Note the different scales for each type.

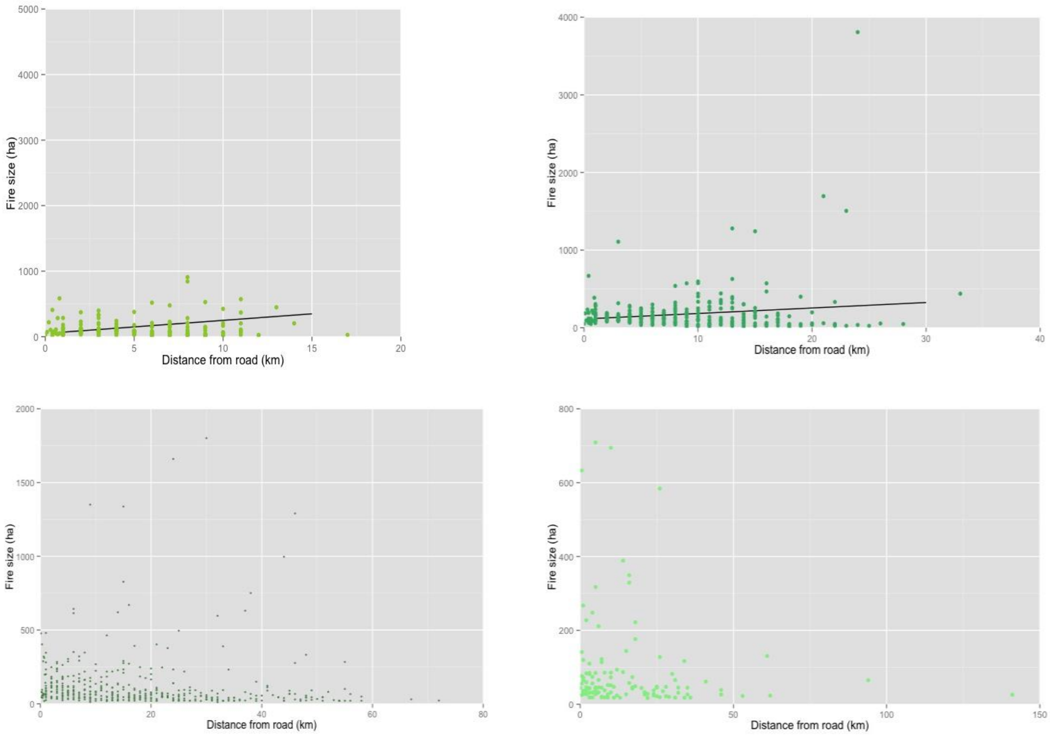


Figure 30. Relationship between road proximity and fire size. Relationships with a regression line were significant (see paper for equation). Note the different scales of plots. Clockwise from top left: broadleaf, mixed, needleleaf, tundra.

Discussion

This study has focused on characterising the fire regime of western Russia, both to determine the extent of variability in fire regime across the whole of the Russian boreal forest, as well as to identify spatial and temporal variability within this region alone. In particular, current perception of Russian fire regime is that it is dominated by high frequency, low intensity fires. Whether this exists in western Russia is uncertain given a relative lack of consideration in previous studies (Soja et al., 2004, Table 1). In particular, central and western forests differ in dominant species composition to areas further East, hence sweeping generalisations on fire regime across this area may miss key differences imposed by forest type and structure (Kharuk et al., 2011). The results of this study indicate, however, that this generalization is at least partially appropriate for the study area. In particular, with some 17,616 fires over the past eighteen years, fires have been of similar frequency to that found in other studies (e.g. Korovin & Isreav, 1998; Ivanova et al., 2010; Kukavskaya et al., 2013). Immediate post-burn fire severity using dNBR also indicates they have had only low-moderate impacts on vegetation (Figure 9). The overall idea of frequent, low-severity fires is supported as peaks in the total number of fires match those in total area burned for each year (Figure 5), suggesting fire impact is controlled largely by the number of fires that occur, as opposed to burn-intensity or severity of effects (Ivanova et al., 2010). In particular, there are no years in which total number of fires is low but burned area is high, suggesting little occurrence of large, extreme fire events in the study area during this period, reflected in the lack of high-severity burns (Figure 9). Results from research in central and eastern Russia find similar figures for fire frequency (see table 1). For example, in a study comparing the fire regimes of Canada and Russia, deGroot et al., (2013), found a mean fire frequency for fires over 200ha of 1441 fires per 100 million hectares, or around 14 per 1 million hectares using the relative scale of this study. The central Siberia study area had similar forest composition to this in that several different forest types were present – including broad-leaved deciduous, southern and northern taiga (mixed and needleleaf-dominant respectively) and forest tundra. With a total of 2,762 large fires in this study, a similar mean annual frequency of 14.23 per 1 million hectares of forested land is found. However, fires in this region, at around 154ha average size, do seem to be larger than those recorded in the east. This fits with the contrast found by Wu et al., (2018), in which fires in their western study area of the North Caucasus biodiversity hotspots were on average four times as large as those found in their eastern study areas (see table 1). This is likely to be because of the influence of understory grassland shrubs within this region, which can help to fuel fires over larger areas (Wu et al., 2018). For forest fire intensity and severity, the majority of studies in central and eastern forests have found predominance of low-intensity surface fires, ranging from 50% in the east (Kukavskaya et al., 2013), 80% across central and eastern regions (Korovin, 1996), to over 90% (deGroot et al., 2013) of fires in southern central Siberia. While the relative amount of stand-replacing (canopy) fires compared to non-stand replacing (surface) fires has not been explicitly measured here, low-moderate severity of effects of all fires on vegetation in this study suggest that here fires are also predominantly low-intensity and likely to be surface fires (Sofronov & Voloktina, 2010; deGroot et al., 2013). However, more recently, other work has recorded distinct north-south spatial variation of this relative pattern, which may explain the variation in the figures stated above (Soja et al., 2004). In particular, a study that considered relative dominance of canopy versus surface fires across the whole of the Russian boreal forest between 2002-2011 found a distinct latitudinal boundary, with forests to south of this dominated by more frequent surface fires, and forests to the north generally experiencing fewer but more extreme stand-replacing fires (Krylov et al., 2014). In particular, the relative dominance of spruce and fir in closed-canopy, dark-needleleaf forests in the north seem to support a highly different regime to the light-needleleaf, larch and pine dominated forests of the south. This is likely to be due to forest-level properties and the aforementioned adaptations, or there lack of, of these species to fire (Wirth, 2005; Krylov et al., 2014). While the results of this study would seem to agree with this latitudinal pattern for southern areas in European Russia, there were few fires detected in the north east of the study area used here and they have not generated significantly different results in terms of fire size or intensity (Table 3). This is may be because a generic MODIS product was used, as opposed to a Russian-based algorithm, which while improving detection of small fires, may have limited capability further north where factors such as snow and rapid growth and senescence periods may have confounded results (e.g. Crevoisier et al., 2007). Hence while this work supports the general east-west pattern of surface fire regime across Russia, it is less clear whether the idea of more severe canopy fires occurring in the north are supported. Overall, however, it is clear current preconceptions of fire regime in Russia, primarily based on studies in central and eastern areas, largely overlook significant large-scale spatial variation in fire regime (Soja et al., 2004; Goetz et al., 2007; Shvidenko & Schepaschenko, 2014). This is important because it is well documented that larger, more severe events are responsible for most of the effects of fire on forest systems, with greater forest loss, carbon release and loss of carbon sequestration in addition to slower rate of recovery and impacts on other ecosystem processes such as soil erosion or flooding (Kajii et al., 2002; Johnstone et al., 2006; Perez-Cabello et al., 2006). Hence, understanding where large fire events occur currently is crucial to understanding and quantifying their effects both now and in the future. It is particularly necessary to address this in climate-vegetation modeling, given that general circulation and other models use current fire regime to predict the consequences and potential feedbacks of climate on the boreal forest system (Soja et al., 2007).

However, while an overview of the type of fire regime experienced across the Russian boreal forest is useful for large-scale predictions, it remains important to note the added layer of spatial and temporal variation within this. In particular, such variation can indicate the relative influence of other factors, both natural and anthropogenic, in determining the effect of fire on forest ecosystems (Cornad & Ivanova, 1997). Spatially, mean fire frequency across the study period is significantly higher in broadleaf compared other cover types (Figure 7). As forest to the south of this area is heavily fragmented by agricultural land, it seems likely this is due to the influence of cropland fires here. Agricultural burning is a common management practice throughout much of Russia, with deposition of black carbon from these fires linked to amplified climate warming of the Arctic (Hall & Loboda, 2017). Studies explicitly considering the impact of agricultural fires in the boreal region are lacking, however they have been found to be a contributing source of anthropogenic wildfire ignition in some areas here (e.g. Shvidenko & Nilsson, 2000; Mollicene et al., 2006). They are also a significant contributor to fire activity in other forests systems globally (Moreira & Pe'er, 2018). This is because cropland fires provide not only potential ignition sources - i.e. drift of burning embers - but also enhance fire-suitable conditions - for example through localized extreme heat (Moreira & Pe'er, 2018). The fragmentation of forest landscapes by agricultural land as well as other anthropogenic activity (e.g roads) can also makes them more susceptible to fire, increasing the extent of vulnerable edge boundaries to allow further penetration of fire into forest stands (Gralewicz, 2008). It would seem that as this region appears to have little temporal correlation with climate forcing (Figures 10-27), agricultural activity could be a major control on fire regime here. However, some caution is needed as the difference could also be due to agricultural fires being mistaken for forest fires (Giglio et al., 2018). While efforts were taken to remove fires identified as burning on non-forested land using LandsSat imagery, there were years when this imagery was not available for the region and identification based on the coarse-resolution of forest cover was difficult. It may therefore be that some cropland fires have been missed and this has artificially inflated the result of mean forest fire frequency. Nonetheless, as cropland fires are known to significantly impact fire statistics in other systems, as well as being major contributors to anthropogenic-induced burns in the boreal forest, the replication of such an effect here is likely.

The overall pattern of when fires occur during the fire season – that is, in later months in the tundra and needleleaf – can also be explained due spatial variation in normal latitudinal climatic differences, with conditions taking more time to become suitable further north (Larsen, 1980). More nuanced is the temporal variation in fire frequency, with significant differences occurring *within* regions according to year and the timing of fires during type-respective normal fire seasons (Figure 6). The inconsistency in annual peaks of fire activity may be explained when examining these with information on temperature anomalies across the region. The peak of fires in mixed forest in 2002 appears to occur after high anomalous temperatures in this area in February-April and again in July and August (Figure 11). It can be seen these are the months in which most of the fires in this region occur (Figure 6). This pattern is repeated for needleleaf forest in 2011 and for northwest tundra in 2018, in which July temperatures are around 4°C above normal, coinciding with increases in fire activity. However, in both the tundra and needleleaf this temperature peak is not sustained – August temperatures for the respective years return to average in the tundra and actually fall below average in the needleleaf region (Figure 11). The continued peak in fire activity in August for both areas therefore seems likely to be due to positive feedback from fire occurrence at the forest-stand level (Shuman et al., 2017). Foremost, fires occurring towards the end of July are likely to increase propagation of fires into August. They are also likely to create conditions more suitable for fire (e.g. embers, localized heat) and thus make fires more likely regardless of higher-level atmospheric conditions (Kajii et al., 2002; Kharuk et al., 2011). However, this must be balanced by the fact that fires occurring earlier in the season can also make conditions less suitable for fire – namely by decreasing or removing fuel load and altering soil edaphic conditions, for example increasing soil moisture by melting permafrost in the tundra (Sofronov & Voloktina, 2010). From the results of this study, it seems that on a yearly basis this negative feedback effect is more significant – with peaks in yearly frequency consistently followed by marked troughs (Figure 7). A similar effect was found in a study on stand-replacing fires across the whole of the Russia between 2001-2012, in particular where the smallest forest loss from these, in 2004, occurred the year after the greatest, in 2003 (Krylov et al., 2014). Vegetation removal not only reduces fuel loading, it also affects albedo by exposing paler bare ground under the forest canopy to reduce heat absorption, reducing suitability of conditions for fire (Chen & Lodoba, 2018). This may explain why there is no consistent relationship of fire activity over this timescale, as peaks – and hence troughs – occur at different times across the study area due to spatial variation in initial climate conditions, which are then positively re-enforced by forest-level feedback. This promotes both a temporal separation from overall climate forcing as well as a spatial separation by cover type. Similar forcing that enhanced the effect of climate extremes was found by Kharuk et al., (2011) in a reconstruction of paleo fire regime in Siberian larch forests. They determined it was the effect of topography on influencing pre and post fire conditions that made these much more common that had previously been documented.

Within the fire season, relative forcing effect of previous fire activity on forest-level conditions and thus successive activity appears to be more nuanced. It seems likely that in mixed forests, early-season fire activity, spurred by earlier temperature and precipitation changes in these regions, removes fuel loading to make later fires less likely. The proportion of birch species within mixed forest is high, and is likely to be the predominant source of fuel for surface fires (Wirth, 2005). As these are not well adapted to withstand fire (but well adapted to colonise after fire), they are likely to be more damaged by fire activity and hence, provide little material for later fires. In contrast, fires earlier in the season in tundra and needleleaf may help to increase likelihood of fires continuing to later months as they support an overall increase from previously much cooler, wetter areas to more fire-suitable conditions in these areas (e.g. Johnstone et al., 2006; Jin et al., 2012). The distribution of fire occurrence in broadleaf throughout the study period is more spurious, however. There appears to be a marked shift in 2009 in when fires occur during the season, from relatively evenly distributed to predominantly occurring within April and May (Figure 6). It is not clear why this occurs. For instance, there does not seem to be any strong temperature forcing according to anomalies during these or proceeding months in 2009 or later years (Figures 18-27). Moreover, predominance of birch would suggest a similar pattern as mixed should be observed. This may therefore indicate, again, the influence of cropland fires on altering fire behaviour within this region. Namely, much cropland burning occurs within spring months (March-May) (Hall & Loboda, 2017), and hence coincides with peaks in fire activity in this region. It is, however, important to note that precipitation conditions are not considered here, but these may be another reason for altering seasonal or annual fire activity across the study area. For example, in the Scots Pine, Larch and deciduous shrub forests of Tuva, Siberia, most burning is associated not only with hot conditions, but also dry periods of low relative humidity, which primarily occur in April and May (Ivanova et al., 2010). It may therefore be that this climate forcing had a dominant effect across southern European Russia during the study period, driving or perhaps exacerbating the observed shift in the broadleaf area.

Forest fire intensity, here indicated by the size of fires as a proxy for total energy output during burning (Keeley, 2009), is also an important statistic with which to evaluate forest fire behaviour. Overall, there does not seem to be any significant temporal variation in mean fire size within cover types (Figure 8, Table 3). Again, this supports that idea that fire regime is predominantly a function of the number of fires, and that this is consistent across the study period. Between cover types, significance was generated in the ANOVA but not the posthoc multiple comparison test. While this is unusual it is not unheard of, and seems likely to stem from variable sample size between each type and year. This is inherent in the nature of fire data, however, and was unavoidable. Data inspection suggests the mean size for broadleaf in 2009 and needleleaf in 2012 may be affecting the ANOVA. In particular, the sample size for needleleaf in 2012 is just three, generating a larger mean even though individually these fires are not outside the range of other years (Figure 8). The spike in mean size for broadleaf in 2010 will be due to this containing the largest fire recorded in the study (48,281 hectares) meaning, again, the mean will be raised despite few other fires being outside the normal range (Figure 8). This is likely to cause the initial ANOVA significant difference, but when testing means under a different null hypothesis in the multiple comparison test, result in no difference. For broadleaf, this large fire event is also likely to raise the overall annual mean compared to other types. Given that the study area is largely composed of six genera at varying levels of dominance that theoretically support similar fire regime (Wirth, 2005), difference between cover types was not necessarily expected. That is, the predominant vegetation in the regions *in which fires occur* – the birch and pine forest of the central and western portions – would not usually support more intense fires that extend further into the canopy or horizontally across a bigger area. This is because of the aforementioned adaptations of these species, which generally limit fuel-loading and prevent fires from becoming larger and more intense. On the other hand, an increase in fire size may have been expected had more fires been recorded in the spruce forest to the north-east of the study area. Here, the extensive, thick, enclosed spruce forest would have facilitated the spread of fires both across the forested area and into the canopy, allowing them to become more intense and much larger (Furyaev et al., 2001; deGroot et al.,2013). The difference for broadleaf compared to other areas may therefore, again, indicate some effect of anthropogenic cropland burning. Namely, the largest fire in the study occurred here and was located in a forest stand in a mosaic of agricultural and forested land. The ignition of cropland fires at several points within this mosaic could therefore have led to them penetrating the stand, promoting fire-suitable conditions and spreading such that they amalgamated into one large fire event (Gralewicz, 2008; Moreira & Pe'er, 2018). Given this occurred in the year after the observed season shift, climate drying may have enhanced this, as observed by Kharuk et al., 2011. Generally, however, it can be seen that fires are all of a similar size. Across the study period, temporal differences in climate or forest-level forcing mainly therefore affect fire frequency, not fire size, and this is a function of forest-level factors that essentially determine how large said fires can become.

Fire severity has also been measured in this study through the use of delta Normalized Burn Ratio (dNBR) to determine post-fire effects on vegetation mortality, and post-fire vegetation recovery rate (Parker et al., 2015). Overall, the results show no temporal or spatial trend in terms of where and when fires had more severe effects on vegetation mortality and post-fire recovery (Figure 9). While there is a clear trend of some recovery for all locations between zero and three years post-burn, none of the locations show any regrowth within in this time (i.e. zero-negative values). Zero to negative values for mixed forest zero-years post-burn may indicate these are from fire scars of previous years, with a known problem of MODIS data being that it can mistake fire scars uncovered by snow melt early in the season as current fires (Crevoisier et al., 2007). Discounting these, no other locations have indication of enhanced vegetation growth (negative values) at three years post-burn. In fact, this only happens in broadleaf just one-year post burn in 2005. Although all cover types would be expected to support the same regime, some level of difference in *recovery* would be expected given the strong contrast of dominant species in growth rate, tree size and wood density (Wirth, 2005; Brandt, 2009). The effects of forest species composition on determining biomass post-fire recovery have been well documented, affecting initial mortality and subsequent competition and succession in regenerating stands (Burkle et al., 2015). In this study, broadleaf and mixed areas would be expected to show faster recovery and enhanced growth post-fire compared to needleleaf and tundra given the ability of deciduous species within the former areas to colonise new ground, exploit favourable post-fire conditions and grow rapidly (Furyaev et al., 2001). As the time frame of dNBR presented here is within a range in which these species could have begun regrowth, it may indicate that fires in these areas were in fact more severe than would be indicated from the USGS classification of the mean dNBR value. In particular, conditions at the time of burning have been shown to have a dominant controlling effect on vegetation recovery, as opposed to environmental conditions in the years post-fire or proximity to unburned areas from which seeds will disperse (Cai et al., 2018). Hence it seems likely lack of recovery for mixed and broadleaf areas could indeed be due to higher intensity fires here. For instance, fires occurring later in the season for these areas not only gives hotter weather conditions but also higher fuel loading, ripe for higher intensity, higher-severity burns. Furthermore, the USGS classification system is fairly generic, designed for use among multiple different ecosystems, hence there may be some discrepancy in the classification given here and what these values actually mean for the boreal forest (Fomacca et al., 2018). Caution is also required given this is a mean statistic. While some burns may actually have experienced recovery, the overall mean may be reduced where there are many fires in one location, but for which data was only available for some in post-fire years (for example, where adjacent LandSat tile had to be used). Furthermore, although conditions at the time of burning may be the greatest influence on vegetation recovery, above ground biomass is by no means the only component of the ecosystem that is affected, and hence must recover, after fire (Keeley, 2009). Recovery of below-ground biomass, as well as successive fire conditions and anthropogenic activity - given this is higher in mixed and broadleaf than other areas - may be having an influence on observed recovery in these areas (Pereze-Cabello et al., 2006). No level of recovery of needleleaf and tundra forest systems would be expected within this time frame. Such systems, with more extreme growth conditions and much slower growth overall, can take anywhere from ten to twenty years to recover pre-fire above ground biomass (Lloyd et al., 2012). While there is a lack of data on fire severity in the tundra, there are certain years where fires seem to be worse in needleleaf-dominated areas. More severe fire effects occur here from burns in 2011, 2014 and 2017. There was a spike in fire activity in 2011 that may be explained by anamolous temperatures during the fire season, however it is unclear whether this made fires particularly more severe or if mean dNBR value has simply been raised relative to the number of fires (Figure 9). In 2014, however, no such spike is recorded yet high anomalous temperatures – around 5°C above normal – also proceed the months of fire activity. This may therefore explain increased severity, with higher intensity but fewer burns. Reasons for high severity in 2017 are less clear, with no correlation to climate forcing (Figure 26). However, all fires occurred late in the season so it may be that high fuel loading enabled them to become more intense. Hence, climate seems to be a dominant control on the intensity of fires across the study period and region, largely controlling the timing of fires throughout the fire season, which then impact forest-level conditions to determine fire intensity. It seems that it is the legacy of immediate post-fire severity that determines recovery rate across the study area.

This study has also found a significant effect of road proximity in influencing the frequency and size of fires in broadleaf, mixed and needleleaf areas, although this effect is not consistent. Namely, road proximity appears to increase the frequency of fires within broadleaf and needleleaf forest, but not in mixed or the tundra (Figure 29). Secondly, road proximity actually appears to limit fire size in broadleaf and mixed forest, but has no effect on size in tundra or needleleaf areas (Figure 30). It may be that low road density and activity in tundra areas limits the amount of traffic and hence human activity that could influence fire frequency and size. For mixed and broadleaf forests, larger fires further away from roads may be because human suppression of fires only occurs within an accessible limit of the road, whereas further away they are not noticed or simply allowed to burn (Campos-Ruiz et al., 2018). The relationships must be viewed with some caution, however, given the nature of the frequency data, where the distance of fires was rounded to the nearest kilometre in order to generate an idea of frequency at a given distance. This may have affected the results by making the distance measurement more categorical rather than continuous. Furthermore, while the relationship is significant, the amount of variation that is actually explained by these relationships (R^2 statistic) is fairly low, particularly in terms of fire size. Nonetheless, there are several mechanisms by which road proximity and road density may affect fire statistics and behaviour within adjacent forest environments. Firstly, as they allow vehicles and people to pass, they increase the likelihood of fire ignition in close proximity to the road due to human carelessness (Campos-Ruiz et al., 2018). Secondly, physical properties along roadways can generate a ‘heat island’ effect to make conditions more suitable for fire, including lower albedo on dark road surfaces, considerable heat absorption of materials such as asphalt, surface friction as vehicles pass, as well as vehicle exhaust fumes (Arienti et al., 2009). Korovin (1996), found that most anthropogenic induced fires in the Russian boreal forest start within close proximity to roads, while over 85% of fires in Siberia are related to some kind of anthropogenic activity (Mollicone et al., 2006). Road density has also been found to increase the likelihood of fire ignition in the Canadian boreal forest by increasing the frequency of fires ignited by lightning strikes, with the authors suggesting this was due to an increase flammable fine fuels near roadways (Arienti et al., 2009). As such, while road proximity or density are included as proxies for the level of anthropogenic influence on fire in general circulation and other models for other forest systems, they have as yet rarely been included in those for the boreal forest (Crevoisier et al., 2007). However, these results, in combination with similar findings from across the boreal forest region, suggest that if we are to thoroughly and accurately understand fire regimes– particularly in the light of where changes to fire frequency and size may affect lives and livelihoods – this is arguably something that should be done with much more regularity for the boreal forest region as a whole.

Overall, it is clear that, across the study area, top-down climate controls and bottom up forest-level conditions interact to control fire statistics and behaviour. In some cases, however, natural and anthropogenic forest-level controls appear to outweigh top-down climate forcing. While this effect is being noted throughout the boreal forest system, it is often crucially overlooked in large-scale predictions of boreal forest fire regimes, particularly when modeled under a changing climate (Soja et al., 2007; Shvidenko & Schepaschenko, 2013). It is therefore clear that these external influencing variables need to be recognised and accounted for more frequently in models or other simulations of future fire regime in the boreal forest, as they are likely to induce significant interactions that may enhance or mitigate large-scale climate forcing, altering ecological processes as well as climate feedbacks.

Given this consideration, the final aim of this study was to understand fire regime of the western Russian boreal forest in light of anthropogenic-induced climate change, which is predicted to have significant impacts on fire regimes throughout the circumpolar boreal region (Kasischke & Turetaky, 2006; Flannigan et al., 2009; Strahlberg et al., 2018). This is especially important because previous studies have already indicated that model predictions are being matched by current observations. For example, across the Russian forest as a whole there was a 29% greater burned area during the 1990s compared to the previous decade (Soja et al., 2007).

Compared to longer-term means, (e.g. Korovin, 1996), there has been a 19% increase in area burned across the entire Russian boreal forest (Soja et al., 2007). In certain areas of the Russian boreal forest, the magnitude of this change has been even greater (Ivanova et al., 2010). It would seem therefore, that there has already been a shift in normal fire regime of the study region compared to older baseline figures. Decadal-scale time frames seem suitable to understand shifts in fire statistics and behaviour related to climate change, and as such, the timeframe of this study can be used to identify change in the most recent decade (2010-2018) compared to the previous (2001-2009). Overall, however, there is no significant difference between the total number of fires or area burned between these decades (Figure 5), and no clear temporal trend of any fire statistic increasing from 2001 to 2018. While some years do generate more fires and related fire effects, this varies spatially and seems to depend more on localized fire-weather and forest-level conditions rather than long term climate forcing. Annual and inter-annual variability in fire statistics and behavior is expected due to variation in forest conditions or fire-weather (e.g. Kajii et al., 2002; Soja et al., 2004; Kharuk et al., 2011). However, the longer-term results of this study seem to go against modeled predictions of increasing frequency and severity of fires in the boreal forest due to climate change (Kelly et al., 2013). They indicate that, so far, this region has not experienced any further significant regime shifts on a decadal-scale compared to 2001. While other factors may be influencing these results for this study, for example the level of anthropogenic influence that affects fires across all years, this finding may reinforce the idea of non-linearity in ecological responses to long-term change (Lloyd et al., 2014; Blume-Wery, 2016). That is, rather than a slow decadal-scale progression in average fire effects it may be that there will be a sudden and pronounced shift in the behaviour of forest fires in this region. While these results are not conclusive, several other works have presented this idea (Chapin et al., 2004; Goetz et al., 2007; Shvidenko & Schepaschenko, 2013). Namely, there are several climate feedback effects in the boreal forest that will exhibit a non-linear and time delayed response to climate change, for example the melt of permafrost and methane release (Serezze & Barry, 2011), albedo feedback (Kuusinen et al., 2012), and growth rates of various species under different growing season conditions (Tei & Sugimoto, 2018). Beyond certain thresholds of climatic change, these processes appear to initiate a non-linear and irreversible response that generally increases warming and, in in the case of the latter, range shifts that cause the loss of some forest species (Chapin et al., 2004). Warming then increases wildfire disturbance, and the inertia present in current feedback responses of boreal forest systems is reduced, overall exponentially accelerating ecosystem shifts due to climate change, including in wildfire frequency and severity (Soja et al., 2007; Ivanova et al., 2010). However, much of our understanding of this process is based on relatively short-term manipulative studies that do not capture natural adaptations of vegetation exposed to climate change over longer periods, hence they can both wildly under and over estimate the scale of long-term effects (Blume-Wery, 2016). Furthermore, there are other time-delayed processes in the boreal forest that may actually exhibit negative feedback on warming and fire disturbance regimes. For example, increased wildfire activity can promote dominance of early successional vegetation within the forest, which helps to stabilize fire regime as these vegetation types do not promote frequent disturbance (Randerson et al., 2006; Soja et al., 2007). It seems these feedbacks are being missed in current modeled predictions because observations are largely based on observed decadal timescales that miss variation in fire and successional cycling that occurs at timescales much longer than this (e.g. Gromtsev, 2002; Kharuk et al., 2011; Shuman et al., 2017). Overall, climate-vegetation feedbacks of the boreal forest are complex. It does not seem there has been any significant effect over the past decade of climate on fire activity within the study area. An analysis over a longer timescale is needed to determine whether baseline rates of fire regime have shifted here, while there is clearly a pressing need to continue to monitor fires and other variables across the boreal forest to determine if, and when, thresholds of potential feedback mechanisms may be surpassed by current warming trends.

Conclusion

This study has attempted to depict fire regime across northwestern boreal Russia over the past eighteen years by quantifying the frequency, intensity and severity of fires and fire effects in this region. The relative influence of different potential controls on fire regime during this time have also been explored to gain better understanding of dominant forces behind temporal and spatial variation. Overall, this study finds there is considerable variation in the pattern of fire occurrence in western Russia, differing spatially within the region and to other regions across the Russia. Namely, interactions between external drivers and internal processes within the forest can both attenuate and enhance the response of fire to dominant-climate controls. This may be why this region has not yet experienced the changes in regime that are already being observed elsewhere in the boreal forest. However, these changes may also be happening at timescales that this study has not addressed. Given the lack of information on historic trends of fire regime in Russia, more work is needed to couple the results of satellite-based observation with ground measurements (e.g. tree ring analysis) to understand regimes in the light of current anthropogenic warming. More pertinently, work is needed to encompass potentially significant forest-fire feedbacks into climate models to generate more accurate predictions of the outcome of warming in this region. In particular, the coming years will be likely to show whether such changes will be the progressive shift that has so far been predicted, or a dramatic and sudden change that will indicate the onset of a new age of fire disturbance in the boreal forest.

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